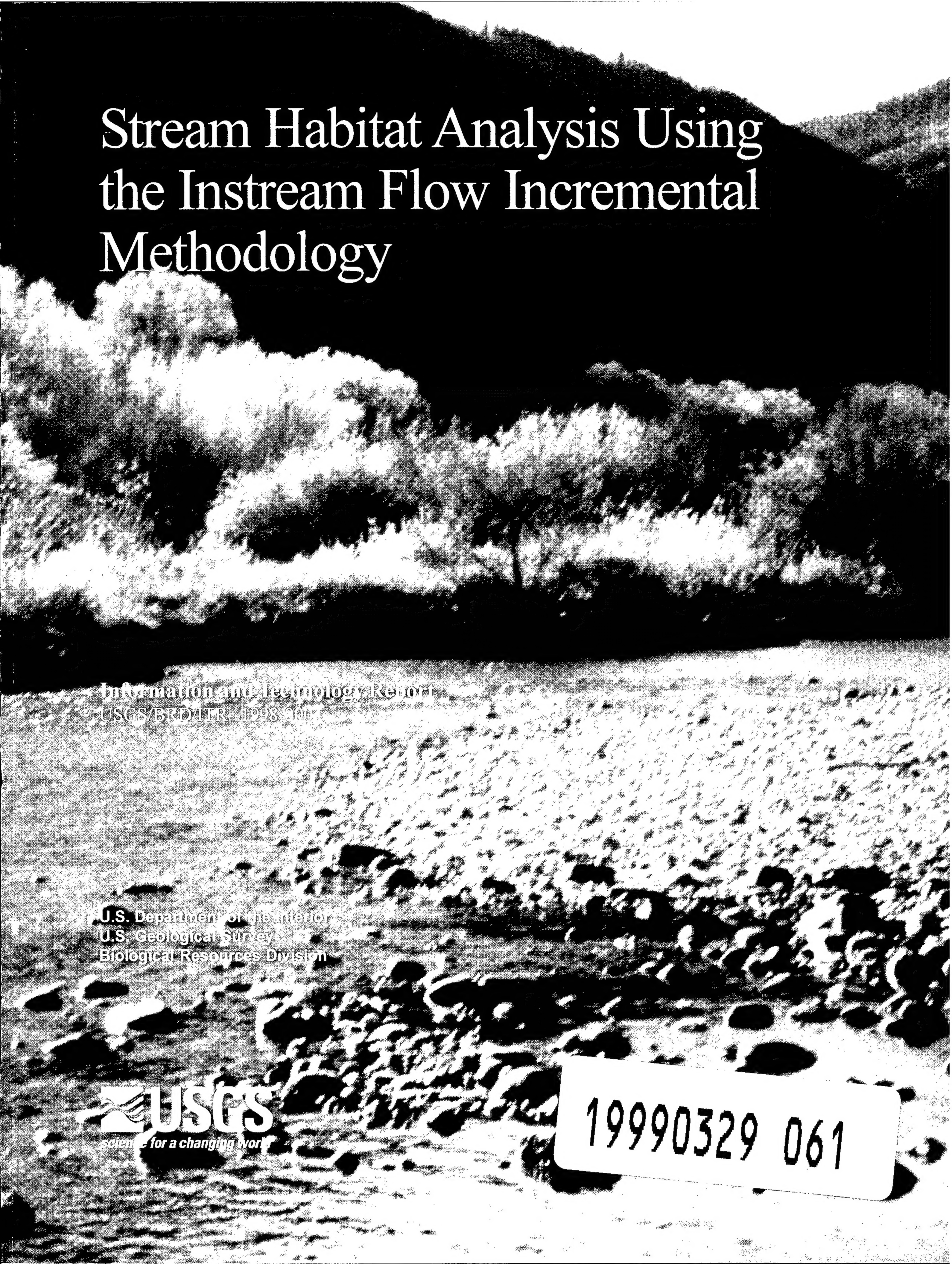

Stream Habitat Analysis Using the Instream Flow Incremental Methodology

Information and Technology Report
USGS/BRD/ITR--1998-0004

U.S. Department of the Interior
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REPORT DOCUMENTATION PAGE	Form approved OMB No. 0704-0188

1. AGENCY USE ONLY (Leave Blank)	**2. REPORT DATE**	**3. REPORT TYPE AND DATES COVERED**

4. TITLE AND SUBTITLE
Stream Habitat Analysis Using the Instream Flow Incremental Methodology

5. FUNDING NUMBERS

6. AUTHOR(S)
Ken D. Bovee, Berton L. Lamb, John M. Bartholow, Clair B. Stalnaker, Jonathan Taylor and Jim Henriksen

7. PERFORMING ORGANIZATION NAME(S) AND ADDRESSES
U.S. Department of the Interior
U.S. Geological Survey
Biological Resources Division

8. PERFORMING ORGANIZATION REPORT NUMBER
USGS/BRD/ITR--1998-0004

9. SPONSORING/MONITORING AGENCY NAME(S) AND ADDRESSES
U.S. Department of the Interior
U.S. Geological Survey

10. SPONSORING, MONITORING AGENCY REPORT NUMBER

11. SUPPLEMENTARY NOTES

12a. DISTRIBUTION/AVAILABILITY STATEMENT
Release unlimited. Available from the National Technical Informartion Service, 5285 Port Royal Road, Springfield, VA 22161 (1-800-553-6847 or 703-487-4650). Available to registered users from the Defense Technical Information Center, Attn: Help Desk, 8722 Kingman Road, Suite 0944, Fort Belvoir, VA 22060-6218 (1-800-225-3842 or 703-767-9050).

12b. DISTRIBUTION CODE
Unlimited

13. ABSTRACT (Maximum 200 words)

This document is intended to update the concepts and ideas first presented in Information Paper 12, the first attempt to describe the Instream Flow Incremental Methodology in its entirety in 1982. This also is to serve as a comprehensive introductory textbook on IFIM for training courses as it contains the most complete and comprehensive description of IFIM in existence today. This should also serve as an official guide to IFIM in publication to counteract the misconceptions about the methodology that have pervaded the professional literature since the mid-1980's as this describes IFIM as it is envisioned by its developers. The document is aimed at the decisionmakers of management and allocation of natural resources in providing them an overview; and to those who design and implement studies to inform the decisionmakers. There should be enough background on model concepts, data requirements, calibration techniques, and quality assurance to help the technical user design and implement a cost-effective application of IFIM that will provide policy-relevant information. Some of the chapters deal with basic organization of IFIM, procedural sequence of applying IFIM starting with problem identification, study planning and implementation, and problem resolution.

14. SUBJECT TERMS (Keywords)
instream flow incremental metholodogy, computer models, decision-support system, integrated habitat simulation and analysis system, riverine habitats, water resource, water quality engineering, fishery biology, social science

15. NUMBER OF PAGES
viii+ 131

16. PRICE CODE

17. SECURITY CLASSIFICATION OF REPORT	**18. SECURITY CLASSIFICATION OF THIS PAGE**	**19. SECURITY CLASSIFICATION OF ABSTRACT**	**20. LIMITATION OF ABSTRACT**
Unclassified	Unclassified	Unclassified	Unlimited

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Suggested citation:

Bovee, K. D., B. L. Lamb, J. M. Bartholow, C. B. Stalnaker, J. Taylor and J. Henriksen. 1998. Stream habitat analysis using the instream flow incremental methodology. U.S. Geologicial Survey, Biological Resources Division Information and Technology Report USGS/BRD-1998-0004. viii + 131 pp.

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Preface

The Instream Flow Incremental Methodology (IFIM) has been around in one form or another since about 1980. The first attempt to describe the methodology in its entirety did not occur until 1982, when *A Guide to Stream Habitat Analysis Using the Instream Flow Incremental Methodology*, Instream Flow Information Paper 12, was published by the U.S. Fish and Wildlife Service (Bovee 1982). As indicated in the first chapter of that work, Information Paper 12 was not the last word on IFIM. We expect that as methodological evolution continues, this report will not be the last word either.

IFIM is one of the most widely used instruments in the world for assessing the effects of flow manipulation on river habitats. It is also widely misconstrued, misinterpreted, and in some cases misused. Our objectives for writing this document were threefold. First, this document is intended to update the concepts and ideas first presented in Information Paper 12. A lot has happened since 1982, and not all of it has been written down (at least not in one place). Second, we needed a comprehensive introductory textbook on IFIM for our training courses. We believe that this document is the most complete and comprehensive description of IFIM in existence today. Third, we think it is important to have an "official" guide to IFIM in publication to counteract the misconceptions about the methodology that have pervaded the professional literature since the mid-1980's. Despite what you read elsewhere, this book describes IFIM as it is envisioned by its developers.

This report is really aimed at two audiences. Our first audience is people who must use technical information to make decisions about management and allocation of natural resources. To these users we hope to provide an overview of IFIM. Natural resources management and allocation decisions are often conducted in a highly charged atmosphere with many conflicting and competing demands. We think that it is important for these decision-makers to know what their options are and what to expect from an analysis that uses IFIM. The second audience consists of the people who design and implement studies so that they can inform the first group. For this group, it is important to understand the strategies involved for producing the exact kind of information that will be needed by the decision-maker. Our intent is to provide enough background on model concepts, data requirements, calibration techniques, and quality assurance to help the technical user design and implement a cost-effective application of IFIM that will provide policy-relevant information. Mostly, however, we intend that this report be used as a guidance document, and not simply read as a literature review.

We have organized the report into five chapters. The first chapter introduces the basic organization of IFIM and explains how it fits with contemporary ecological philosophies. The next four chapters describe the procedural sequence of applying IFIM, starting with problem identification, then moving to study planning and implementation, and ending with the problem resolution phase. If you learn nothing more about IFIM, you should learn that it is far more than a collection of computer programs. This report does not discuss any other instream flow methods besides IFIM, nor does it speak to the historical, theoretical, and philosophical underpinnings of IFIM. For this type of background information we refer you to Stalnaker et al. (1995).

We are indebted to the dozens of former students of the IF 250 course who provided comments and suggestions for improvements of earlier drafts of this book. We also acknowledge the thorough peer reviews provided by Jim Terrell, Thom Hardy, and an anonymous reviewer. We have done our best to incorporate their ideas into the book, and we thank them for all their efforts. Finally, we thank the people involved in the final publication of this book. Duane Asherin served as the executive editor for the Midcontinent Ecological Science Center. Jennifer Shoemaker organized the publishing process for us, Dora Medellin edited and formatted the text, and Dale Crawford was responsible for many of the more imaginative line drawings. We also acknowledge the staff of the National Wetlands Research Center for their assistance in preparing this document for publication.

March 1998

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Abstract. This document is intended to update the concepts and ideas first presented in Information Paper 12, the first attempt to describe the Instream Flow Incremental Methodology in its entirety in 1982. This also is to serve as a comprehensive introductory textbook on IFIM for training courses as it contains the most complete and comprehensive description of IFIM in existence today. This should also serve as an official guide to IFIM in publication to counteract the misconceptions about the methodology that have pervaded the professional literature since the mid-1980's as this describes IFIM as it is envisioned by its developers. The document is aimed at the decisionmakers of management and allocation of natural resources in providing them an overview; and to those who design and implement studies to inform the decisionmakers. There should be enough background on model concepts, data requirements, calibration techniques, and quality assurance to help the technical user design and implement a cost-effective application of IFIM that will provide policy-relevant information. Some of the chapters deal with basic organization of IFIM, procedural sequence of applying IFIM starting with problem identification, study planning and implementation, and problem resolution.

Keywords: hydrology, instream flow, negotiation, decision making, management, instruction, problem identification, problem resolution

Chapter 1. What is IFIM?

IFIM is an Interdisciplinary Problem-solving Tool

The Instream Flow Incremental Methodology (IFIM) is a decision-support system designed to help natural resource managers and their constituencies determine the benefits or consequences of different water management alternatives. Some people think of IFIM as a collection of computer models. This perception is understandable because IFIM is supported by an integrated habitat simulation and analysis system that was developed to assist users in applications of the methodology. However, IFIM should be considered primarily as a process for solving water resource allocation problems that include concerns for riverine habitat resources. IFIM was developed under leadership of the U.S. Fish and Wildlife Service by an interdisciplinary team of scientists drawn from Federal and State resource agencies and academia (Trihey and Stalnaker 1985; Stalnaker 1993). The first decade in development of this methodology focused on integration of numerous techniques developed from water resource and water quality engineering, fishery biology, and social science (Stalnaker 1982).

Historically, instream flow determinations often consisted of arguing for flows to maximize the amount of microhabitat for a single life stage of a high-profile fish species at a few isolated spots in a river. Decision-making within the context of IFIM has matured to the point that flows recommended for any time period or scenario are subject to scrutiny and evaluation. To deal with such intense examination of alternatives, one of the most powerful features of IFIM is the mechanism for experimenting with various river regulation schemes. Water budgets allocated for fish production and policy decisions for storage and release from reservoirs are becoming the realm of the natural resource manager (Waddle 1991). For these reasons, the authors of this text strongly advocate an interdisciplinary team approach to the use of IFIM. The reader is advised that the remainder of this text is heavily influenced by the philosophy of interdisciplinary teams.

As IFIM technology continues to evolve, it is advancing from an impact assessment tool to a water planning and management tool for setting policy on river regulation. Reservoir operation and water routing models, when coupled with habitat time series analysis of IFIM, allow numerous alternative water release schemes to be compared (Harpman et al. 1993; Waddle 1993). By gaining a place at the water management table, the resource manager may now be responsible for managing a portion of the water supply that is dedicated to instream flow needs. The ecologist-as-water manager must acquire an understanding of the hydrologic conditions of the watershed, including the historical patterns of water supply (drought and flood cycles and period of recurrence), the water rights held and priority of use, and the typical patterns of delivery for the major water users in the river system being managed.

IFIM is a Modular Decision Support System

IFIM is composed of a library of linked analytical procedures that describe the spatial and temporal features of habitat resulting from a given river regulation alternative (Fig. 1-1). The methodology is adaptive in that components can be combined to fit specific needs. One of the unique characteristics of IFIM is the simultaneous analysis of habitat variability over time and space.

It is commonplace to operate at several temporal and spatial scales during an application of IFIM. A hierarchical classification system, similar to the one developed by Hawkins et al. (1993), is used in IFIM to describe habitat characteristics at various scales (Fig. 1-2). It is especially important for practitioners to have a good understanding of the various spatial scales used to define habitat in IFIM. Data collection and analysis at a small scale are combined with larger scale measures to build habitat-flow models at the larger scale.

Three macrohabitat-level stratifications may be used in an IFIM analysis: drainage basins, networks, and segments. The largest habitat unit is the drainage basin, which may range in size from tens to thousands of square kilometers. A network usually consists of two or more sub-basins but may encompass an entire drainage basin. The segment is the smallest macrohabitat stratum and is considered to be the fundamental habitat accounting unit used in IFIM.

The next smaller scale is mesohabitat. Mesohabitats can contain many microhabitats, but they are typified by a common slope, channel shape, and structure. Pools and riffles are familiar mesohabitats. The length of a mesohabitat type is commonly about the same order of magnitude as the width of the channel. Mesohabitats can be subdivided into microhabitat components, which range in area from less than 1 to several square meters. A microhabitat is defined as a localized area of stream having relatively homogeneous conditions of depth, velocity, substrate, and cover.

How the components of IFIM are assembled and combined depends on the nature of the problem and the objectives of the study. Microhabitat can be integrated longitudinally with macrohabitat variables of water chemistry and temperature to develop functional relationships between total habitat and discharge for the entire segment. Channel structure is analyzed at the macrohabitat level when it pertains to its effects on water quality or temperature, or when the objective is to maintain an existing morphology.

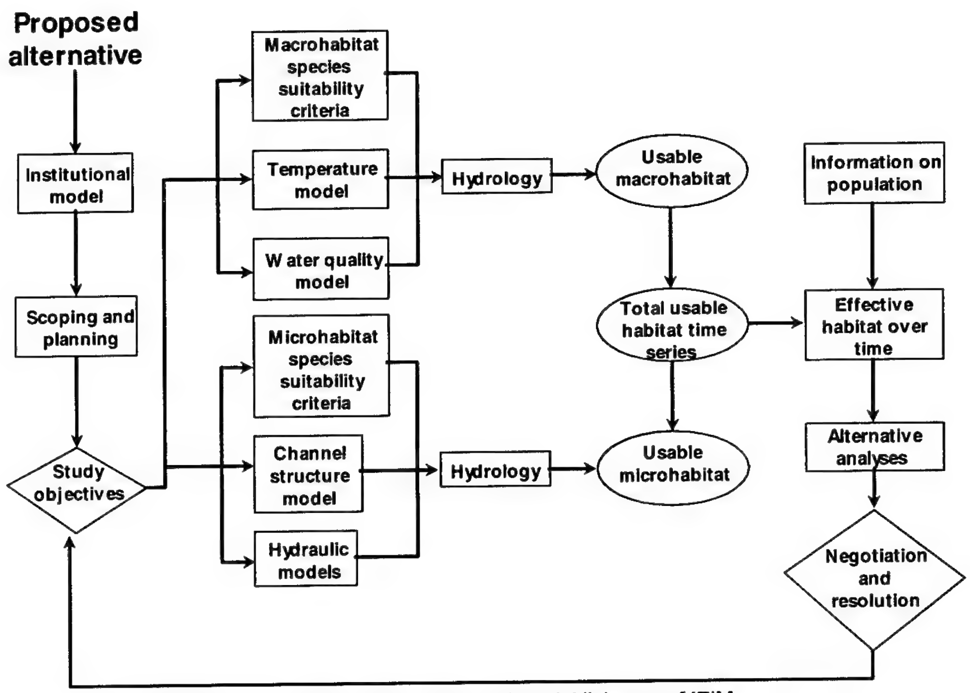

Fig. 1-1. Schematic diagram of the components and model linkages of IFIM.

Channel structure may also be analyzed at the microhabitat level to evaluate changes in microhabitat availability resulting from a change in channel morphology. Reservoir operations and stream network analysis link stream hydrology and total habitat for individual or multiple segments. These engineering models provide the means for conducting temporal analysis of habitat dynamics and, in rare instances, fish population dynamics (Waddle 1992).

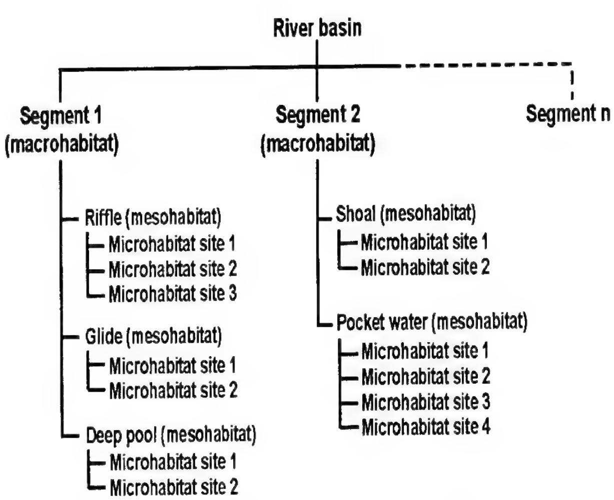

Fig 1-2. Stratification of habitat units used in IFIM analyses arrayed in order of decreasing scale.

The effects on microhabitat resulting from peaking hydroelectric operations and the reevaluation of large storage reservoir operations have recently elevated the instream flow management issue in the United States to the stream network and river basin scale (Lubinski 1992; National Research Council 1992; Hesse and Sheets 1993).

IFIM is Grounded on Ecological Principles

Karr and associates (Gorman and Karr 1978; Karr et al. 1986), in developing the Index of Biological Integrity (IBI), suggested that human-induced impacts to river systems fall into five major categories: flow regime, habitat structure, water quality, food source, and biotic interactions. Table 1-1 recasts these mechanisms as factors to consider when identifying potential impacts resulting from a disruption of one or more pathways. IFIM's modeling approach has been influenced by this view of river system impacts, and models have been developed to fit within this paradigm.

Flow Regime

During construction of large storage reservoirs and massive withdrawal systems in the western United States during the 1950's and 1960's, the concern of natural resource managers was focused on evaluating changes in flow regime. Thus, a hydrologic component is evident during several steps of IFIM (e.g., Fig. 1-1). Hydrology drives the temporal analysis within IFIM because the amount of habitat in a segment at any given time is related to the streamflow

Table 1-1. Variables important to the biological integrity of river ecosystems, organized according to pathways affected by human-induced alterations (modified from Karr [1991]).

Flow regime	Habitat structure	Water quality	Food source	Biotic interactions
Discharge	Habitat diversity	Nutrients	Algal production	Exotic species
Water depth	Siltation	Thermal regime	Energy input	Endemic species
Water velocity	Bank stability	Turbidity	Particulate organic matter	Candidate for threatened and endangered species
Flood frequency	Cover	Salinity	Aquatic invertebrates	Hybridization
Flood magnitude	Woody debris	Dissolved oxygen	Terrestrial invertebrates	Population structure
Drought frequency	Channel sinuosity	pH		Competition
Drought magnitude	Riparian vegetation	Toxins		Species richness
Flow variablity	Habitat connectivity			Predation
				Trophic structure

at that time. One of the most important considerations in designing an IFIM analysis is to determine the appropriate period of record and time-step to be used to describe habitat variability under baseline and alternative flow regimes.

Habitat Structure

Habitat structure is quantified at the microhabitat scale but aggregated to the mesohabitat scale in IFIM. The Physical Habitat Simulation System (PHABSIM) (Milhous et al. 1989) is an integrated collection of hydraulic and microhabitat simulation models designed to quantify the amount of microhabitat available for a target species over a wide range of discharges. PHABSIM combines empirical descriptions of the structural features of the channel, simulated distributions of depth and velocity, and habitat suitability criteria for the target species. This combination reveals a functional relationship between streamflow and the area of microhabitat available for the target species, per unit length of stream. Hydraulic simulation models allow accurate prediction of hydraulic variables (e.g., depth, velocity, width) for discharges that were not or could not be measured. Such simulations allow the analyst to evaluate the duration and timing of inundation of the aquatic-terrestrial transition zone (Junk et al. 1989). It is also possible to describe relations between streamflow and habitat structure in more general terms, such as habitat diversity or richness.

Understanding the influence that human-induced activity has on habitat structure remains one of the most neglected research areas within stream ecology (Hill et al. 1991). The loss of side channel, backwater, and edge habitats has been one of the primary reasons for the decline of many riverine species (Hesse and Sheets 1993). This concern is manifested by the efforts of the U.S. Army Corps of Engineers to restore floodplain-channel connections along river corridors that have been severely impacted through channelization projects. The classification of mesohabitat types and associated species assemblages within major segments of river underscores the importance of adequately describing and managing channel morphology as a component of river management.

There has been considerable empirical research focused on the protection of existing channels (Rosgen et al. 1986; Stalnaker et al. 1989) and the restoration of floodplain habitats (Hesse and Sheets 1993). Recent research in the western areas of the United States and Canada focuses on flushing flows as part of river management regimes for the purpose of flushing silts and sands from interstitial spaces among gravel and cobbles in trout and salmon streams (Reiser et al. 1989a). This research should provide algorithms suitable for computing the magnitude and timing of flow pulses for flushing fine sediments from river reaches below large storage reservoirs. At present, one of the foremost problems in riverine systems is the relationship between geomorphology and ecology. The techniques available to predict channel responses to changes in flow regime or sediment transport are crude, at best.

Water Quality

Temperature and water quality are macrohabitat components of IFIM. IFIM studies generally incorporate water quality models in common use by the water resource or public health agency of the region (Bartholow 1989; Thornton et al. 1990). Guidance documents for evaluating and recommending temperature regimes in conjunction

with IFIM applications have been prepared for some important fish species (Armour 1991, 1993).

Food Energy Source

To date, flow-related models for evaluating the food base in streams have been restricted to simulations of microhabitat area for use by benthic macroinvertebrates in streams inhabited by trout and salmon. Such models are based on occupancy of different substrate and velocity conditions by various species of aquatic insects (Sprules 1947; Needham and Usinger 1956; Minshall 1984; Gore 1987). Microhabitat use models for aquatic macroinvertebrates generally follow procedures outlined by Gore and Judy (1981) and were recently demonstrated by Jowett (1993) to account for a significant amount of the variation in brown trout production among 89 trout streams in New Zealand.

Biotic Interactions

Of the five pathways listed by Karr (1991), the biotic pathway has been most neglected and offers much promise for further development. Interspecific competition as a consequence of flow management has thus far taken the form of examining the amount of habitat overlap between trout species (Nehring and Miller 1987; Loar and West 1992). Careful examination of simulated historical temperature and flow patterns for a stream reach may provide evidence for hypotheses to explain the observed dominance of one species over another in the reach. Unfavorable temperature during spawning and incubation, unfavorably high velocities during fry emergence, or great overlap in preferred rearing or resting space during critical periods all may tip the balance in favor of one species over another. Further research is needed for developing habitat models based on community structure. One concept is that of habitat-use guilds of fishes, as discussed by Leonard and Orth (1988) and Bain and Boltz (1989). Recently, research along these lines has been conducted in coastal and piedmont warmwater stream systems in the southeastern United States (Bain and Boltz 1989; Freeman and Crance 1993).

The initial focus of instream flow studies using IFIM is to understand the dynamics of habitat change under historical flow conditions in the stream system under study. This text presents analytical procedures designed to relate habitat dynamics and hydrology. These procedures provide information compatible with four current concepts of stream ecosystems: (1) the longitudinal succession concept introduced by Shelford (1911), elaborated on by Trautman (1942), Burton and Odum (1945), and Vannote et al. (1980); (2) habitat segregation and the importance of habitat patchiness and habitat boundaries in resource partitioning (Chapman 1962, 1966; Wiens 1977; Schlosser 1982, 1987); (3) the flood pulse concept introduced by Junk et al. (1989); and (4) the biotic responses to stochastic processes (Grossman et al. 1982; Schlosser 1987). In dynamic stream environments, the integration of all these concepts is necessary to sort out the relative importance of deterministic and stochastic processes to the community being studied (Schlosser 1982; Gelwick 1990; Strange et al. 1991).

IFIM is an Evolving Methodology

Hundreds of IFIM applications as well as numerous critiques and calls for improvement appeared in the 1980's (Mathur et al. 1985; Morhardt 1986; Shirvell 1986; Orth 1987; Scott and Shirvell 1987; Gore and Nestler 1988; Lamb 1989). Under the auspices of the American Fisheries Society (AFS), two surveys were conducted in 1981 and 1986 of all State, Provincial, and Federal fisheries agencies in North America, including numerous IFIM users. The survey resulted in a compilation and summarization of the extent of the use of IFIM across North America, and a prioritized statement of research needs (Reiser et al. 1989b). Identified priority research needs were to (1) define the relation among flow, habitat, and fish production; (2) validate and test the relation between IFIM habitat output and fish production; and (3) develop new methods for determining flow requirements in warmwater communities and where species habitat information was lacking. Since the AFS survey, research and development have focused on items 1 and 2, and have involved chinook salmon, rainbow trout, brown trout, and smallmouth bass. Recently, an intensive research program was initiated on warmwater stream communities in the southeastern United States (Bain and Boltz 1989; Freeman and Crance 1993). Armour and Taylor (1991) listed the component computer models of IFIM and identified those needing improvement and testing.

In the future, we believe linking the dynamics of habitat to the dynamics of fish populations and community characteristics will become increasingly important. This linkage cannot be accomplished in the absence of population data from the stream under study. A minimum amount of population information will need to be collected on-site to calibrate effective habitat simulations. Typically, some monitoring takes place on tailwaters that are being intensively managed for sport fishes. Creel surveys of the angler's catch, or age and growth studies are additional low-effort means of monitoring the status of a fish population.

Habitat time series (Trihey 1981; Milhous et al. 1990), in conjunction with effective habitat analysis (Bovee 1982), allow the manager to determine if there are associations between weak or strong year-classes and patterns of habitat constriction or abundance of available habitat in the simulated history of the stream. By adjusting the ratios of habitat required among the life stages, the manager can develop a time series of effective habitat that corresponds with population trends and patterns of year-class strength, calculated growth histories, and other anecdotal information on the population status.

With any collection of models, error is introduced through input measures and model simplification of natural systems. Calibration and parameter adjustment are essential steps in applying models to accurately predict changes to the natural system resulting from a change in system inputs (Fig. 1-3). The biologist on an interdisciplinary team must ensure biological realism and modeling accuracy by insisting that the habitat suitability criteria used in the habitat models be relevant and accurate. The engineer on the interdisciplinary team must ensure the accuracy of physical models through calibration and testing of the hydraulic, water quality, and water routing models.

For intensively managed streams and populations, a new generation of habitat management models is evolving (Hagar et al. 1988; Cheslak and Jacobson 1990; Stalnaker 1994). These modeling efforts involve the integration of contemporary fish population models with spatiotemporal habitat models. Such modeling requires a considerable knowledge of the fish population, including seasonal and annual mortality rates, seasonal patterns of movement within the stream network, and estimates of habitat carrying capacity for each life stage (Williamson et al. 1993).

Simulation models provide the negotiator with the kind of information necessary to have a portion of the water supply dedicated for instream benefits (sometimes known as environmental water). Natural resource managers are increasingly becoming water managers involved in annual and seasonal decision-making on how to release environmental water to the river. To become skilled at this type of planning, the resource manager must learn how to forecast water supplies and the runoff characteristics of the river basin. For example, following uncontrolled catastrophic events such as severe droughts or floods that greatly reduce aquatic diversity, the allocated water might be banked in storage and smaller amounts delivered downstream until the demand for habitat increases through reproduction and immigration. Banking water in such a manner may reduce the risk associated with a series of droughts.

By examining simulations of the general state of the fish population and the forecasted water supply, the manager can call for storage or release of water much as traditional water users do. Intensively managing supplies allows for much more efficient uses of water for instream purposes than do the constant minimum releases that have been standard practice over the last two decades. Waddle (1992) demonstrated that releasing reservoir water as needed to provide fish habitat resulted in a more robust fish population than a constant minimum release of the same volume of water. A larger population of fish could be supported over 10 to 20 years with the same annual storage budget

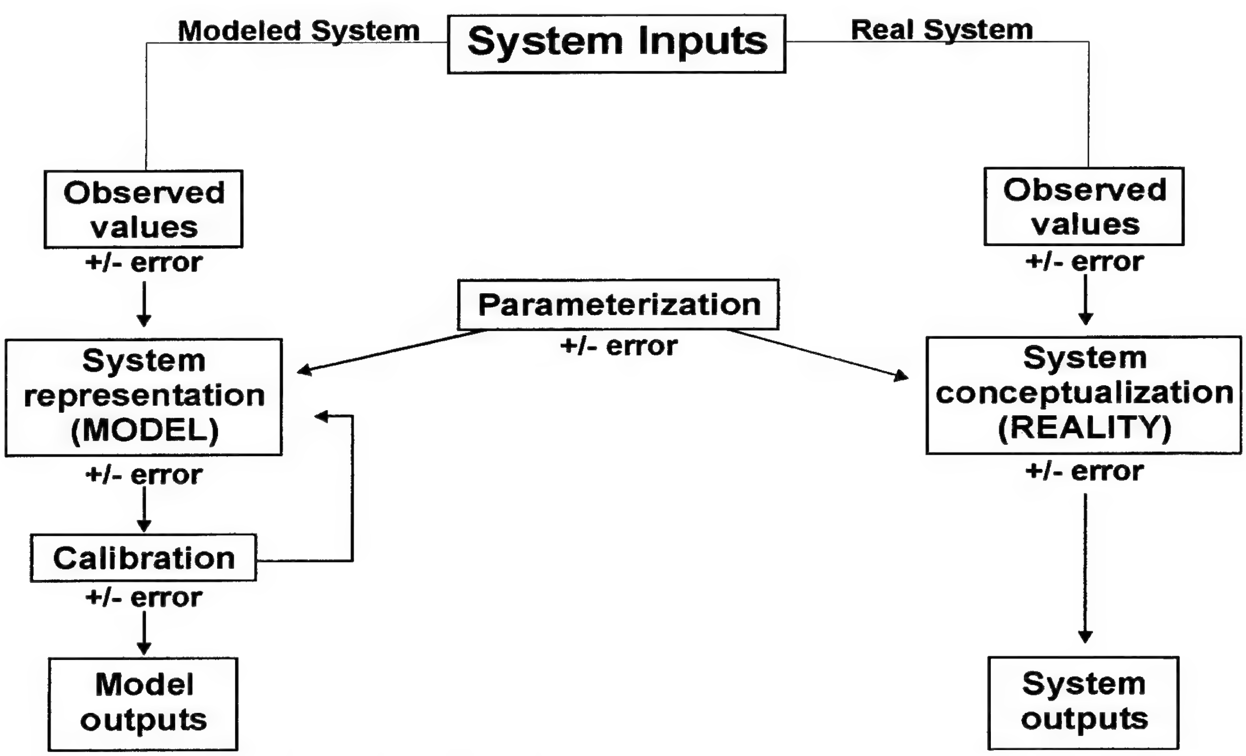

Fig. 1-3. Sources of error when models are used to represent real systems, illustrating the calibration step essential for accurate and realistic model predictions.

when managed to match the population needs, as opposed to being delivered as a constant minimum flow.

Modeling tools provide access to the field of environmental water management. In this field, the natural resource manager must learn to deal with other disciplines and use scientific tools in a social setting very different from descriptive biology. We see a paradigm shift away from belief in protected minimum flows toward managed instream flows within a multiple-use context.

IFIM is a Process

The final perspective of IFIM is that of a process (Fig. 1-4) consisting of four interrelated activities or phases: problem identification and diagnosis, study planning, study implementation, and alternatives analysis/problem resolution.

Problem identification and diagnosis consists of two principal components: (1) a legal and institutional analysis to define the problem setting and the probable context of its resolution, and (2) an issues analysis that identifies concerns of the various stakeholders of a problem and the information that will be needed to resolve the problem.

Study planning involves a comparison of information needs with information already available. The difference between needed and available information is the basis for the study plan. During the formulation of a study plan, an interdisciplinary team must agree on study objectives and deadlines, appropriate models and data requirements, levels of temporal and spatial detail, roles and responsibilities, products and milestones, and project budgets. Study planning should also develop a common understanding of the analytical approach that will be used for evaluating alternatives.

Study implementation involves data collection, model calibration, and verification of model input and output. Quality assurance is necessary at every step in study implementation to ensure that the information produced by IFIM's component models is as accurate and realistic as possible. Without trustworthy data, it is difficult to accurately compare alternatives that might be proposed during the next phase.

During alternatives analysis/problem resolution, an agreed-on set of baseline hydrologic conditions provides the essential point of reference. All parties to the decision process may then have their preferred alternatives compared with the baseline conditions. The group can collectively examine all alternatives for their effectiveness, physical feasibility, risk of failure, and economic considerations. Problem resolution is accomplished through negotiation and compromise, based on the evaluation of competing alternatives. Interdisciplinary teams composed of various stakeholder groups can derive solutions through iterative

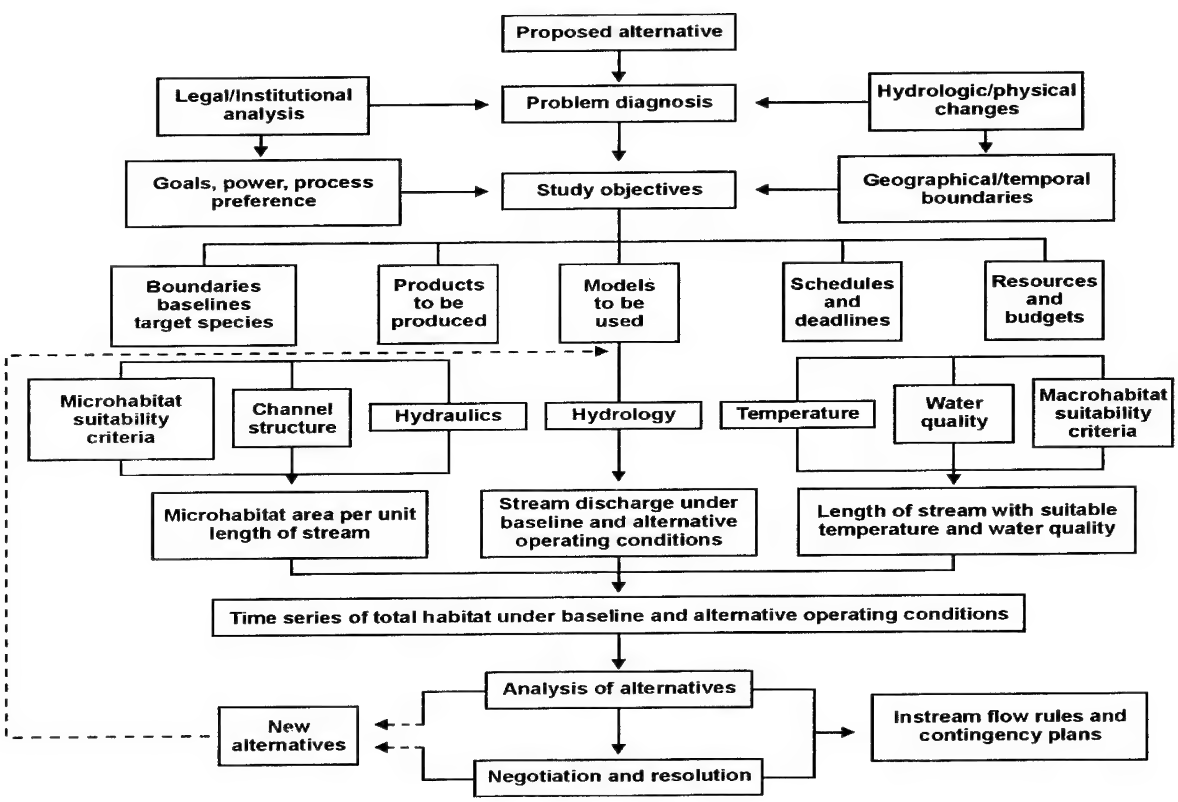

Fig. 1-4. Schematic diagram of activities and information flow involved in an IFIM study.

problem-solving to achieve some balance among multiple and often conflicting uses of water.

IFIM provides a framework for decision-making in the realm of multiple-use water management. It involves interdisciplinary problem solving, incrementalism, bargaining, and craftsmanship. Lamb (1989), in discussing IFIM technology, paraphrased an essay by Schlesinger (1968):

> Certainly careful use of methodology can accomplish something—hopefully a great deal. Nonetheless the resistances to the application of systematic and rigorous analysis in a highly politicized environment are sufficient to make even the stoutest heart grow faint . . . .

Lamb closed his discussion by cautioning the practitioner, agency representative, or entrepreneur to be well advised that "there is no such thing as one-best-way." No tool, no matter how sophisticated, will produce *the answer*. Consequently, IFIM users are expected to use their best judgement in organizing their logic and documenting their assumptions. It is our intent to provide insight and guidelines for conducting an IFIM study in the context of a multiple-use river system setting.

Chapter 2. IFIM Phase I

Problem Identification and Diagnosis

Investigations of alternative management options involving riverine habitats are commonly referred to as instream flow studies, but this term may be a bit constraining for IFIM analyses. An instream flow study implies that the only solution to a riverine habitat problem is to change the streamflow. As you learn more about the parts of IFIM, you should come to realize that changing the streamflow is an effective way to manage riverine habitat, but it is certainly not your only option.

Projects that involve IFIM can be rather intimidating, especially for stakeholders who have never been involved in an IFIM project before. There are many misconceptions about IFIM, so the early stages of an IFIM study may be plagued by mistrust and poor communications. Misgivings about individuals and groups sometimes result in posturing and defense of positions rather than reasoned discussions of the issues. As an investigator using IFIM, two of your most important functions are to encourage healthy communication and foster trust among the stakeholders whenever possible. Uneasiness and uncertainty among stakeholders is understandable whenever something new is proposed. In the absence of facts, people often assume the worst with respect to new proposals that may affect them. One of the most important steps in dispelling misgivings among the stakeholders is to hear their concerns and to address those concerns as part of the analysis. In an application of IFIM, this important first phase consists of two parts: a diagnosis of the institutional setting of the problem and an analysis of the issues that will be forthcoming from various groups of stakeholders. These two steps are necessary precursors to bounding and scoping the plan of study.

Legal and Institutional Analysis

Professionals in natural resource agencies often find themselves in negotiations over environmental impacts and mitigation. Negotiation is to be expected whenever IFIM is used for instream flow assessment. Some people believe IFIM is a tool that provides the one best answer to any instream flow problem. This picture of IFIM implies that deciding upon a recommendation is merely a matter of knowing how well the IFIM analysis was conducted. If IFIM was soundly conducted, the resulting recommendation should be clear and incontrovertible. This vision is incomplete for two reasons. First, the image of a static technique ignores the primary reason for using IFIM. The methodology was developed for the express purpose of providing a common language and rationale for assessing the viability of competing operating scenarios. Although the techniques embodied in IFIM are based on established scientific knowledge, the methodology is designed to avoid suggesting the one-best-answer. Second, the power of IFIM lies in the ability of users to agree on objectives, alternative models, appropriate sampling regimes, and data interpretation. IFIM is predicated on negotiation.

Another way of saying this is to acknowledge that IFIM is as much policy analysis as it is fishery biology or hydrology. Policy is defined as purposive action taken by public authorities on behalf of or affecting the public (Hofferbert 1974). Policy analysis is an investigation of policy either to determine likely outcomes or to understand how a decision was made. Conducting a policy analysis means the user of IFIM engages in a complicated modeling process that is intended to help decision-makers. Helping decision-makers in this way has been labeled "speaking truth to power" (Wildavsky 1979). Arriving at the truth is a matter of balancing conflicting objectives, data limitations, and available time to produce recommendations decision-makers can use to establish an appropriate management alternative. Sometimes the users of IFIM are also the decision-makers and sometimes the decision comes from a source outside the study process. No matter where the decision resides, the IFIM user must prepare a study that satisfies the needs of the policy-making process.

Problem Solving

The most successful IFIM practitioners are problem solvers. Acting as a problem solver for instream flow issues implies knowledge of many different disciplines and an orientation toward the human side of policy analysis; that is, understanding what people need to know to make a decision. The primary skill for a problem-solver is understanding that not everyone shares the same perspective. Because the way people see things often determines their beliefs, it is vital that instream flow analysts grasp the perspective of potential decision-makers as well as the viewpoints of other analysts with whom the task of making instream flow recommendations is shared. Perhaps the most serious error a problem solver can make is to suppose that everyone sees a problem through the same eyes.

Perception is not merely a function of eyesight. Human perception is most profoundly a product of training and experience. How you feel about different areas of the country, how you feel about computers, and how you feel about instream flow are all outgrowths of the way you have been conditioned to see the world. For example, after studying IFIM you might ask yourself how anyone can talk about streamflow in terms of a single year-round minimum flow; it is all in a person's perception of the issue! Another aspect of perception is the role of values. Shared values are often accompanied by shared perceptions, and the opposite is

also frequently true. Differences in perception mirror deep-seated differences between individuals. Moreover, the way you see a problem may change depending upon the situation. Your view may change when you approach an instream flow question from the position of an applicant as opposed to a regulator.

Examples of Perception Problems
Public signage is a common place to find double meanings. Here are two examples:
EAT HERE AND GET GAS
FOR RESTROOM USE STAIRS
These signs are useful only when we share the perception of their authors.

Another skill of a problem solver is the ability to understand the concept of rationality. One perspective of a rational approach has been labeled "rational-comprehensive" (Lindblom 1959). According to this view, a good decision is one that is rational and comprehensive. Rational means following a logical and step-wise process, while comprehensive means that the decision-maker has considered all the facts, potential alternatives, and potential outcomes. Because no one can actually process information in this comprehensive way, it is common for casual observers of human behavior to conclude that people are not rational. The model of a rational person is Mr. Spock on the television program *Star Trek*, for whom all thinking is strictly logical. Social scientists contend that people are indeed rational, although not in the cold, logical Spock model. People are rational in that they reasonably seek to satisfy their own interests. One way this rationality is manifested is in the customary human practice of making decisions one piece at a time and basing the current choice on a precedent (Simon 1957; Lindblom 1959).

Relying on precedent means using analogy to classify problems into types. These types might be referred to as problem settings. Each setting represents a set of complimentary and conflicting needs. When a decision-maker approaches a familiar setting, it is possible to rely on precedent to predict the needs of all the players. Precedent gives boundaries to the current problem in a way the rational-comprehensive approach cannot; our experience tells us what to expect and makes the current problem understandable. Once you know the problem setting you also know the suite of solutions that have been used in the past. This knowledge allows you to reach a decision similar to one that met your needs in the past. Categorizing problems by analogy gives you a rational shorthand for decisions.

Levels of Analysis

Each problem setting can be divided into separate contexts. It is useful to think of three contexts: individual, institutional (group), and systemic (intergroup). The individual context refers to a person-to-person interaction. The question is "how does he or she decide?" When you must convince a single person, the focus is on how to present information to that individual. The institutional context focuses on the human group. The question is "how do we all decide?" When you are working as part of an institution, the spotlight of your attention is on the dynamics of group formation and group decision-making. The systemic context is reflected in bargaining among organizations. The question is "how do collections of organizations decide?"

Pigeonholing problems by their context (individual, institutional, or systemic) is referred to as the levels of analysis approach to problem solving (Singer 1969). The levels of analysis approach acknowledges that there is a vast difference in decision-making between individuals and collections of human groups. A strategy that is appropriate to bargaining with a single person is often ineffective in the inter-group context. For example, there is a difference in context when an individual is negotiating to buy a used automobile and when an individual is representing an agency in an interagency regulatory consultation.

Three Contexts for Problem Solving
Individual: One-on-one
"How does a person decide?"
Group: An assembly of people
"How does the group decide?"
Intergroup: Several groups
"How do factions decide?"

One of the useful aspects of the levels of analysis approach is that it allows a professional to dissect and diagnose problems. Making an instream flow recommendation involves all three levels of analysis. At the individual level, you will be required to convince colleagues about objectives, techniques, and data interpretation. At the institutional level, you must form an interdisciplinary group of investigators who can function effectively to carry out a study. In fact, any one analyst may be a member of several groups. At the systemic level, you represent your organization in an interagency bargaining session. Social scientists typically focus on only one level of analysis at a time, but as a practitioner you must be able to diagnose problems that reflect behavior at all three levels.

Individual Level of Analysis

At the individual level of analysis, focus is on personality and personal style. If you can imagine that every individual possesses a personal map (a complex set of preconceived notions) of particular problems, it will be easy to conclude that people must spend considerable effort at map building, map maintenance, and map revision. In fact, it is so difficult to revise a personal map that people tend to resist that kind of change. The idea that people resist changing their mind set is described in the theory of cognitive dissonance (Festinger 1957). Cognitive dissonance

Reasoning by Analogy

Generalizations from one analysis level may act as supporting statements or corollaries to other generalizations from a different analysis level, but often they cannot be combined into a single true statement. Yet the tapestry that may be formed from these complementary threads . . . yields rich knowledge. (Whicker et al. 1993)

refers to the tendency of people to hold on to their personal map for as long as possible. The theory depicts individuals as working so hard on map maintenance that there is little inclination to perform map revisions. Once a personal cognitive map is constructed, a great deal of emotional capital must be invested to change it. Consequently, the individual works to maintain cognitive harmony and in so doing actively avoids information or approaches that challenge the validity of the personal map. At the individual level, bargaining over instream flow recommendations is a process of developing and challenging cognitive maps. The most successful analyst will find a way to integrate new information into another person's existing map without requiring the individual to perform major map revision.

Cognitive dissonance, however, is not the whole story. Layered on top of the tendency to avoid changes in the cognitive map is personality type or decision style. Most of the ideas about personality type are based on the work Carl Jung reported just after the beginning of the 20th century. Most employees of State or Federal government, and many private sector employees as well, have had the opportunity to assess their own personality type using a Meyers and Briggs personality sorter. Because building an IFIM study is based on negotiation, it is important to understand the way different personalities approach problems, digest information, and form conclusions.

Institutional Level of Analysis

At the institutional level, an organization's internal behavior is determined by the concepts of incrementalism (Lindblom 1959), agency culture and standard operating procedures (Allison 1971), and peer pressure (Janis 1972). It is possible to understand much of the internal workings of an organization by observing how that organization has responded to similar circumstances. Lindblom's (1959) famous essay on incrementalism demonstrated that each new problem is likely to be resolved with a decision only slightly different from past decisions. The reason for this is safety and security. If a past decision worked, there is little reason to take a chance on a completely new approach. The tendency to repeat past actions is compounded by the effects of agency culture. It is difficult for even a visionary leader to impose new directions on an agency with an established culture and formal processes (Allison 1971). As powerful as those two factors are, peer pressure is the cement that ensures consistency. Janis (1972) described studies of groups of decision-makers. After a while, the groups developed a shared perspective and actively worked to maintain that perspective, even in the face of contradictory information. These groups were typical in that they developed unspoken norms that were quietly and effectively imposed on dissident members. These factors combine to produce a consistency of organization behavior that can be relied upon in planning a negotiation.

Systemic (Intergroup) Level of Analysis

At the intergroup level, behavior is conditioned by an organization's mission as expressed in authorizing statutes and the positions that have been worked out in past interorganizational negotiations. Organizations send forth representatives who possess a knowledge of the agency's position and the positions other agencies and groups have taken in similar negotiations. Every negotiator is knowledgeable of the agency's sources of power, scope of responsibility, and authorities. Representatives also know the positions taken in similar cases, means commonly employed to gather and analyze data, and the official relations with other organizations (Lamb 1980). All of these elements work together to define an organization's behavior. Knowing what behavior to expect will tell you the strategy and tactics an organization is likely to employ. Knowing the likely negotiation behavior of others allows you to design an effective bargaining strategy (Clarke and McCool 1985; Wilds 1986).

When you attend an interagency instream flow bargaining session you represent your organization. The most common predictor of the outcome of such a negotiation is power (Lamb and Doerksen 1978). Power in interagency natural resources negotiations may have many forms, but chief among them are the powers of the organization itself (Burkardt et al. 1997). For example, you may have a magnetic personality but work for an agency that has only a minor interest in the conflict. No matter how hard you try in such a situation your effectiveness will be limited by your organization's position in that negotiation.

Institutional Analysis

The term institution means those legal, political, administrative, and agency "structures and processes through which decisions are made . . . " with respect to policy, projects, regulations, licenses, and/or permits (Ingram et al. 1984). These structures and processes include more than just laws and regulations. Laws and regulations are merely the formal guideposts for decisions. Structures and processes also include informal elements. Of primary concern is understanding how agencies, interest groups, and other interested parties are likely to behave in a negotiation. Institutional analysis can be divided into two processes: understanding agency perceptions and evaluating policies. The phrase agency perceptions refers to the way agencies view the process of negotiation, and policy evaluation

A Case Study: The Terror Lake Project

The Federal Energy Regulatory Commission (FERC) is responsible for granting licenses under the Federal Power Act (16 U.S.C. 792 *et seq.*) for non-Federal entities to operate hydroelectric projects. This act provides, in part, that FERC may condition a license to protect the public interest and must balance both power and nonpower considerations. In the Terror Lake case these considerations included both instream uses of water and terrestrial habitat values. As is typical of FERC licensing activities, the applicant for the Terror Lake license was required to consult with State and Federal natural resource agencies to work out license conditions.

The Terror River is located on Kodiak Island in Alaska. The river flows out of Terror Lake into the Gulf of Alaska. Both the river and lake are within the boundaries of the Kodiak National Wildlife Refuge. The river is an important resource for hydroelectric power to serve the people of Kodiak Island. Among the many issues involved in the Terror Lake Project were instream uses of water for fish habitat. The Terror River helps sustain a commercial run of Pacific salmon, which are also a prime food source of Kodiak brown bears. One element of the negotiations focused on instream flows to maintain this fishery. Protection of habitat for the Kodiak brown bear is one of the main purposes of the Kodiak National Wildlife Refuge. An integral part of that habitat is its flowing water resources.

The project was designed to raise the level of Terror Lake and divert water through a penstock and power station into a different river basin. It was first planned in 1964 and a Federal Power Commission (FPC) license was applied for in 1967. The FPC preliminary permit expired and a new license application was filed with the FERC in 1974. During 1976-81, the applicant—Kodiak Electric Association (KEA)—and many interested agencies negotiated in stages over what should be studied and what conditions should be included in the license. During the environmental impact assessment process, the interested parties most often employed either a cursory institutional analysis or simply failed to study the political elements of the problem.

Although the Terror Lake Project was successfully licensed and built, in the beginning there was a poor understanding of the problem-solving process. This lack of understanding arose, in part, because the parties did not conduct an institutional analysis. Spiro (1970) pointed out that "method, or approach, in politics as in everything else, is of supreme importance." The use of institutional analysis as part of the standard approach to instream flow questions helps the parties avoid some common pitfalls. Five conclusions from the Terror Lake project highlight the lessons that might have been available from an institutional analysis:

1) In any negotiation, all the parties should record understandings, agreements, time-tables, and other milestones. This was not done by the Terror Lake Project negotiatiors and, consequently, disagreement remained over the details.
2) It is essential that monitoring be a part of a negotiated solution.
3) A mediator may be an asset in a negotiation. FERC staff was involved in mediating the Terror Lake Project negotiations.
4) The parties to a negotiation face uncertain solutions if they allow decisions to be made by distant agencies and commissions. The best resolution comes from parties closest to the project.
5) The tendency to see each faction (either an agency or group of individuals) as a unified entity seems to encumber negotiation. For example, parties in the Terror Lake negotiation had little sense at the beginning of existing organizational divisions or dynamics. The result was that parties were often perplexed by an opponent's stance on issues.

Policy Engineering

The technology for environmental impact assessment has grown at an exciting pace. But even with improvements, new problems arise faster than solutions. Several analysts have written that the "people" skills of applied scientists have not kept pace with technology (Margolis 1973; Ingram et al. 1984). Because of rapid change, a threefold problem has manifested itself. First, environmental protection questions are increasing in number. This increase comes both in the number of individual problems and in the types of challenges. Second, while techniques for understanding and managing these problems develop rapidly, the new methods lag behind the problems. There are more threats to the environment than there is knowledge to deal with them. Examples of this situation can be seen in recreation management and hazardous waste. Third, a new awareness is developing which acknowledges the shortage of skills to manage problem solving. Problem solving abilities are the institutional analysis skills that lead to successful legislation, judicial decisions, policies, and negotiations. Absent the ability to perform institutional analysis there is inadequate guidance for choosing appropriate technologies (Lamb and Lovrich 1987; Lamb 1993).

Part of the challenge for engineers, biologists, chemists, and atmospheric scientists in conducting effective institutional analyses lies in a misunderstanding of the geometric nature of institutional relationships (Kane et al. 1973). Ingram et al. (1984) commented that "while technicians are willing to acknowledge that institutional factors must be considered, they are not at all clear about just what needs to be taken into account." Behn (1981) said there are two kinds of applied scientists: "policy analysts" and "policy politicians." The policy analysts use technology to resolve scientific questions but leave to decision-makers the choice of what the results mean. Because technical analysis is not planned for policy questions and is not displayed in a policy-relevant manner, the ultimate decisions are often the result of misinterpretation and misunderstanding. Policy politicians, on the other hand, are interested in the process through which their analysis becomes a part of policy. They try to see analysis through to its practical application. Behn (1981) believed there were too few policy politicians. Ingram et al. (1984) argued that even the policy politicians do not seem to know how to analyze an institutional problem. Ranney (1976) wrote that what is needed in these situations is "political engineering." He called on social scientists to develop the practical tools necessary to solve problems. He referred to these political techniques as "empirically derived general principles of . . . institutional behavior."

refers to choosing a course of action and helping others reach decisions.

Agency Perceptions

Understanding agency perceptions is a two-step process. First, you need to carefully assess your own mandates, positions, and relative influence. An accurate understanding of your organization's policies, resources, skills, and influence is critical to successful bargaining. Consequently, careful attention should be given to defining your position and developing a strategy for interaction. Second, you need to assess the position, influence, and resources of other parties. In the Terror Lake process, the Kodiak Electric Association (KEA) was particularly skillful at assessing the other parties and their experience demonstrates that such analysis is essential. Even though KEA did not set out to systematically analyze the positions and political resources of other agencies, they did hire a politically astute consultant who brought negotiation experience and valuable intuition to bear on the bargaining (Olive and Lamb 1984). Indeed, all the parties in such a complex undertaking need to be able to assess the background and strategies of their counterparts.

Policy Evaluation

For an agency or utility that anticipates involvement in environmental negotiations, it seems evident that an established capability to analyze institutional processes would be beneficial. The FERC staff proved adept at this type of institutional problem scoping in the Terror Lake case. The staff had the ability to see the problem as a large mosaic. Thus, they could visualize how the negotiation should progress as well as identify the stakeholders' likely behavior. The ability to conduct such policy evaluation is manifested in a more holistic view of the problem and more coordinated responses.

Institutional Analysis Method

Effective negotiation requires accurate and practical institutional analysis (Nierenberg 1973). Too often, such analysis is either descriptive instead of behavioral or ideological instead of objective. To overcome these shortcomings, a model of institutional behavior has been developed for use by agency analysts, interest groups, and decision-makers. This model is known as the Legal-Institutional Analysis Model (LIAM; Lamb 1980; Wilds 1986).

The model postulates sources of agency power and primary decision strategies. This knowledge is used to predict behavior for each primary strategy. These strategies are referred to as roles. Participants playing these roles are found in most water negotiations. They include advocates who demand change in the traditional decision processes; guardians who seek to protect the status quo (especially by relying on time-tried decision processes); brokers who seek to manage decisions through tradeoffs and bargaining; and arbitrators who endeavor to make objective, court-like decisions.

The institutional model guides you in determining which roles are present and weighing each role according to various power factors. You then use each party's historical behavior pattern as a guide to future conduct. By following this procedure you may examine the institutional context, determine whether technology or politics controls certain decisions, and fashion negotiations in a way that helps solve problems.

Organizations struggle to develop knowledge of their own position and the position of the other participants. This knowledge consists of data on the sources of power, recollection of previous positions on similar problems, information on the means commonly employed by other parties, and a personal relationship with specific individual representatives also involved in the arena. The outcome depends on how skillfully you use power, knowledge, and information in dealing with adversaries (Lamb 1976; Doerksen and Lamb 1979). Discovering your own power, knowledge, and information and that of other parties in a negotiation is the purpose of an institutional analysis. The LIAM involves four steps: determining roles, describing context, calculating power, and assessing strengths and weaknesses.

Determining Roles

In bargaining situations, organizations play certain roles with remarkable consistency. Golembiewski (1976) noted that "agencies or their units tend to develop distinctive styles, much as individuals do. These styles help determine the policies that get adopted and the decisions that are made, and both policies and decisions strengthen as well as derive from the organization style."

Reliance on roles means that the actions of each organization are likely to be consistent with the organization's traditional behavior (Wildavsky 1975). Roles are determined by mission, support groups, and specific problems at hand (Olive 1981). Based on research into the instream flow decision-making process, standard roles in natural resource management have been split into two categories: allocators and activists (Lamb 1980). The first step in conducting an institutional analysis is to determine which roles are being played in the negotiation. Allocators ultimately decide how benefits are distributed. Activists challenge the rules, appeal to the allocators, and try to win as much as they can.

Allocator roles. Particular kinds of decision contexts indicate which of the allocator roles (arbitrator and broker) are likely to be present. Arbitrators are organizations that have statutory authority to establish management plans or regulations, establish the guidelines for preparing plans, or direct the implementation of the plans by others. They rely on data collected by others and make authoritative allocations after hearing evidence from all sides. In the Terror Lake Project, the FERC served as an arbitrator because of its court-like decision process.

Brokers are agencies that have the ability to facilitate bargaining. They are in a position to help or hinder the planning and implementation process. In bargaining they tend to rely on cost-benefit analysis, mechanisms for controlling resource allocation, and to some extent, political considerations. The latter is important because of the nature of the agencies' support groups. The strategy seems to be to guide decision-making in order to retain the balance-of-power (Beckett and Lamb 1976; Lamb 1976, 1980). In the Terror Lake Project, the Office of the Secretary, U.S. Department of the Interior played this role by encouraging the various parties to keep bargaining (Olive and Lamb 1984).

Activist roles. Activists are the direct competitors in a negotiation. At one end of a continuum are found the advocates: agencies that call for a change in the status quo approach to natural resources management (Wildavsky 1975). These agencies often must react to threatened changes in the management context. They may rely on "crusading" and data analysis to advance their position. The distinguishing factor is that the advocate tends to resist the imposition of development- or economic-progress philosophy on a project (Lamb and Lovrich 1987). In the Terror Lake Project, the U.S. Fish and Wildlife Service's Division of Ecological Services was one of the advocates.

At the other end of the continuum are guardians: agencies that attempt to protect themselves and their constituency from interference. They are interested in protecting against challenges to their routines or plans. They guard against change in management practice or project design (Wildavsky 1975). These agencies may prefer some legal or political strategies in bargaining, such as interest group consultation or public participation. Guardians prefer routine procedures because they are well established and time tested, and because their support groups can be influential in the existing decision roles associated with these procedures (Beckett and Lamb 1976; Lamb 1976, 1980). In the Terror Lake Project, the KEA was the guardian. The guardian in this case experienced significant differences with the advocates (such as the Division of Ecological Services and the Division of Refuges) and the arbitrator (FERC).

These activists design their behavior to accommodate the presence of an arbitrator or broker (Olive 1981). Different activities are pursued within the activist roles depending on the situation. Advocate agencies, for example, often develop alliances with arbitrators because the arbitrators rely on advocates for information and opportunity to act. Guardians often pursue holding actions or seek to use their constituency to show injury from an advocate's initiatives.

Agency roles, though typically quite consistent over time, may change. Usually, if an agency has been a guardian in one type of problem, that agency will be a guardian when the same type of problem reappears. This holds true for all roles, but an agency may be a guardian on one issue and an arbitrator on another. For example, the U.S. Forest Service emphasizes a guardian role when it comes to land management policies. They seek to protect the status quo. But on issues regarding rights-of-way, it becomes an arbitrator, using the objective analysis procedure in granting permits. The U.S. Fish and Wildlife Service, Division of Ecological Services, is an advocate agency regarding U.S. Army Corps of Engineers projects or endangered species actions, but the Division of Refuges is a guardian on its refuges.

Describing Context

Two arenas of policy-making exist in which the natural resources management game may be played. However, the players are always advocates, guardians, brokers and/or arbitrators. One arena is known as **distributive politics.** In this arena success is determined by such things as "fair share," "base," and "legitimacy" (Ingram 1972; Wildavsky 1975). In this game, winning means ensuring that all legitimate parties (i.e., parties with a rightful place in the negotiation) have some minimum base (i.e., share of the resource). As the total potential rewards grow, each party can be assured of its fair share (i.e., proper increased payoff over the base amount). A natural resource problem can be thought of as a game to divide the rewards from any project, regulation, or management scheme so that all legitimate players get some benefits. This is the traditional way to do business in natural resources management (Ingram and McCain 1977).

The other arena of policy-making is **regulatory politics.** Typical decisions in this arena are based on objective, reasoned analysis. The idea is not to divide a pie, but to decide who is "right" and who is "wrong." That is, the actions of arbitrators are directed to clear-cut decisions. This choice is related to facts, circumstances, and precedents. Obviously, behavior of the parties is different in this arena than in distributive politics. The courtroom, with its strictly defined adversarial procedures, is a typical regulatory venue.

It is possible to visualize these two arenas as opposite ends of a continuum; distributive politics on one extreme and, on the other, regulatory politics. In each of these arenas a different allocator role dominates. Fig. 2-1 illustrates how the broker dominates the distributive politics arena while the arbitrator dominates in regulatory politics (Olive 1981). In the example of Terror Lake, the decision was first worked out in the distributive arena and then finally decided upon in the regulatory arena (Olive and Lamb 1984).

Historically, advocates have been left out of distributive politics or believe they have been treated unfairly there. To continue the game metaphor, visualize all the parties in distributive politics working with traditional allies and ignoring advocates so that the rules rarely allow the advocate even to play. As a result the advocate has enjoyed the

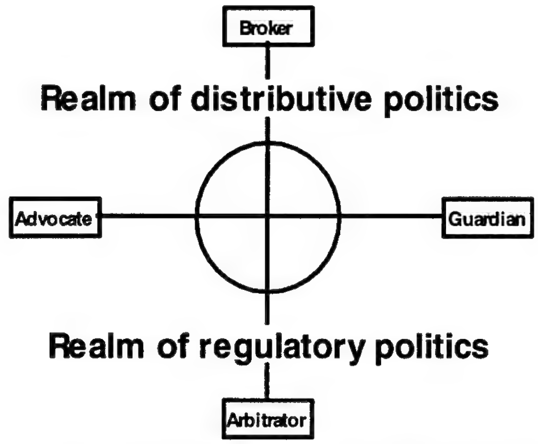

Fig. 2-1. Role map for the Legal and Institutional Analysis Model (LIAM).

most success in the regulatory arena. Advocates sometimes gain access to the distributive arena from the regulatory arena. For example, when an organization is denied a seat at the bargaining table, it may appeal to a court to ask for an injunction. Under such a court order, the bargaining process would stop until the court decides who should be included.

As a consequence of this history, advocates usually try to push the problem into regulatory politics. Because guardians have historically been legitimate parties in distributive politics, however, they try to force the parties into the distributive arena. Thus, the tensions in the model are compounded by parties trying to move the game to the two different arenas (Olive 1981).

There are three key elements of success in distributive politics which advocates must master: expertise, compromise, and constituency. The advocate usually has expertise and, given the environmental movement, this is often enough for admission to the arena (Ingram 1972). Beyond providing a ticket for admission, however, the game rarely allows expertise to play a controlling part. Results come from compromise. Although advocates are getting better at compromise, this is a skill that brokers have long mastered and guardians often use. As a result, compromise is the common language of distributive politics. Everyone understands that decisions are made through compromise, and most parties also know that constituency is the basis for arranging compromises. The skill of compromise can be learned, but constituency is an organization's basic source of strength.

In the distributive arena all legitimate parties receive some reward. That behavioral fact underscores the difference between distributive and regulatory politics. The regulatory arena can be a zero-sum game; in playing regulatory politics, there is always a chance to be judged wrong. Thus, in choosing an arena, parties trade frustration against uncertainty.

Calculating Power

A quick glance at Fig. 2-1 should not lead to the conclusion that all parties assume the most extreme roles. Indeed, it seems to depend on circumstances. In particular, the way a role is played seems to be based on the power of the agency and its stake in the problem. In this game, some parties are second-string and some are first-string. Relative power makes a big difference in behavior (Lamb and Doerksen 1978).

Agency power can be distinguished by three elements: knowledge, resources, and again, constituency (Wilds 1986). Knowledge is defined as subject area expertise, the ability to process information, and the understandability of an agency's expressed opinions and policies. Assessing the degree to which agencies hold knowledge can be important in the prediction of negotiation behavior. For example, in the Terror Lake Project the U.S. Fish and Wildlife Service possessed important technical expertise while the KEA expressed itself in commonly understood language (Olive and Lamb 1984).

Resources refer to statutory authority, physical control of the resource, legal management responsibility, financial backing, and available personnel. These elements of resource power are fairly straightforward. Two others are more oblique: the frequency and intensity of involvement. Frequency of involvement is the idea that experience in negotiation is an important power resource. The more experienced organization has evolved routines, attitudes, and personnel directed to successful bargaining. Intensity of involvement is a measure of how close the issue at hand stands in relation to agency mission. The closer to agency mission, the more intensely interested is the agency. Intense interest is a resource because it contributes to an organization's attention to the issues under consideration. An intensely interested organization is one that cannot be easily ignored.

Constituency is the power brought to an organization by its supporters. Here, constituency refers to either political support (elected officials) or public support (support by organized interest groups). Public support is a measure of how well constituency groups are paying attention to the issue at hand, how cohesive the groups are on the issue, and how astute they are in presenting and advocating their views. The power inherent in these three elements determines the strategies an agency will pursue and its vigor in the pursuit. Power is a predictor of success in bargaining (Burkardt et al. 1997). Fig. 2-2 illustrates the roles and power of the parties involved in the Terror Lake project consultations.

Assessing Strengths and Weaknesses

The final step in an institutional analysis is to predict strengths and weaknesses. To do this, you must turn again

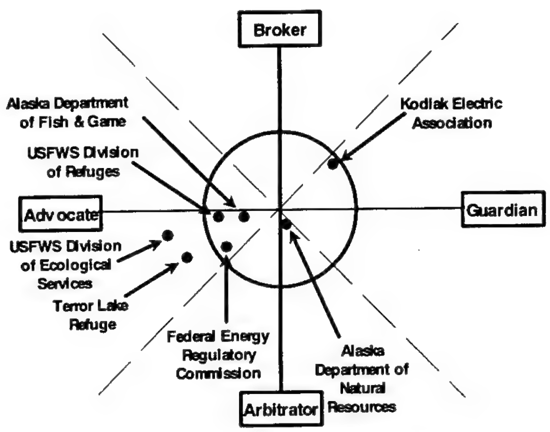

Fig. 2-2. LIAM role map for the participating agencies in the Terror River case study.

to the elements of power. After you understand the likely roles to be played, the context of the negotiation, and the raw power of each organization, you should be able to describe a basic negotiation strategy. The question then becomes one of tactics. Where should you offer to assist another party? On what subjects should you assert yourself? An analysis of strengths and weaknesses can help answer these questions.

To construct an analysis of strengths and weaknesses you first array the elements of power. For example, using the elements of resource power illustrated in Table 2-1, describe the power of each party to the negotiation. Match the power of your organization to that of the other parties. You should find an opportunity for action where another party is weak and you are strong. Conversely, you should expect negotiation pressure when you are weak where another is strong. The action you may take in a position of strength depends on the circumstances. A potential tactic is offering to share some of your strength with a weaker party. For example, if your power includes adequate staffing you could offer to take on some tasks on behalf of other parties that are understaffed.

The Connection Between LIAM and IFIM

The idea behind LIAM was to help IFIM users choose the kind of technical information they would need in a negotiation. Lamb (1993) described how different political situations require appropriate technologies. For example, a conflict in which highly charged values clash, the parties are likely to end up in court. Such a situation requires a technology like IFIM, which is scientifically sound and can depict the results of complex alternative flow regimes. LIAM can provide a picture of negotiations that allows you to decide if you face such a problem. Fig. 2-3 shows a negotiation scenario that is likely when extremely held positions diverge over values. The nature of this conflict may be deduced from the extreme differences on the advocate-guardian continuum and the added presence of an intense arbitrator.

The extreme divergence between advocates and guardians in Fig. 2-3 indicates a fundamental difference in the values of the two groups. The presence of a strong arbitrator almost certainly means that this problem will be resolved in a court (or court-like) setting, rather than in a negotiation. An extensive technological analysis is called for because as a negotiator, you will be facing an intense, far-ranging inquiry about methods, results, and recommendations. You will need the knowledge that will allow you to flexibly approach alternatives over the course of a protracted dispute.

In a less intense conflict, during negotiations that are not so charged (Fig. 2-4), the parties perceive the problem similarly, desire compatible outcomes, and may feel comfortable with less extensive technical analysis. Finally, the conflict you face might be more a matter of low public awareness than differing values or disagreements among agencies with similar management objectives. This low-intensity situation (Fig. 2-5) may involve a decision-maker (broker) who is interested in making trade-offs based on clear-cut relationships between variables. If this is the situation you face and no further conflicts are anticipated, you may

Table 2-1. Elements of resource power in natural resource negotiations as described in Lamb and Doerksen (1978) and illustrated for use with Legal Institutional Analysis Model (LIAM) in Wilds (1986).

Power element	Examples of strong power
Statutory authority	A clear legislative mandate to act
Physical control of resource	Ability to control water flow
Legal control of resource	Designation as implementing agency or land manager
Political support	Legislators favorably disposed to organization
Public support	Organized, cohesive constituency
Fiscal resources	Adequate budget focused on issue
Personnel	Adequate staffing focused on issue
Frequency of involvement	Experiences with similar issues
Intensity of involvement	Issue close to organization's mission

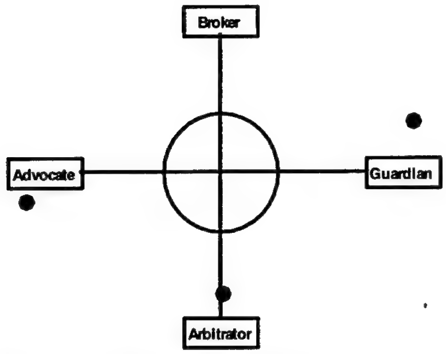

Fig. 2-3. Display from LIAM showing strong divergence on goal preference and the presence of an arbitrator. Goal divergence is indicated by the wide separation between advocates and guardians. An arbitrated settlement is suggested by the presence of a strong arbitrator on the broker-arbitrator axis.

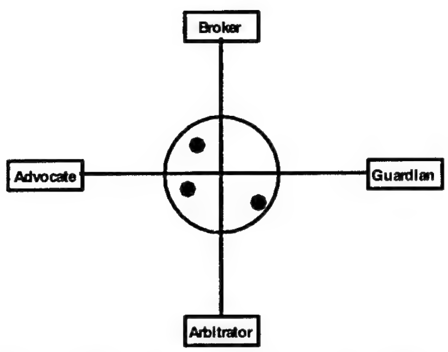

Fig. 2-4. Display from LIAM showing similar goal preferences among all stakeholders. In this case, the separation of stakeholders is small on both axes, indicating a noncontroversial settlement.

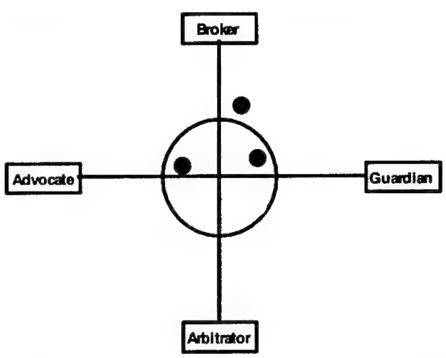

Fig. 2-5. Display from LIAM that indicates a brokered decision because all of the negotiators are categorized into the realm of distributive, rather than regulatory, politics.

choose to use IFIM but rely on expert knowledge to answer more detailed questions that might arise. Lamb and Lovrich (1987) discuss how to choose a strategy for this sort of problem.

Government decisions made through interagency and interest group bargaining appear to some commentators as the wrong means to decide public policy. For example, Lowi (1969) decried the tendency of government to respond only to interest group pressure because it undermines public authority and is necessarily conservative of the status quo. Lowi demonstrated that it is false to assume that everyone actually gets a chance to participate in public decisions. He argued that even though the decision-making process is participative, interested citizens are held out of the discussions because they are not represented by organized groups. Nevertheless, interorganizational bargaining over policy seems to be the approach most Americans believe in. Negotiation is certainly the most common form of decision-making about instream flow issues.

Decision-making on instream flow issues can be likened to a game in which the parties pull and haul on one another until some kind of a bargain is struck. Yaffee (1982) showed that even where policy is supposed to be closed to compromises, intergroup bargaining controls implementation. The Terror Lake Project is a good example of intergroup bargaining. Many different groups representing a wide range of interests participated. The consensus seems to be that the process was successful (Olive and Lamb 1984; K. Bayha, U.S. Fish and Wildlife Service, Anchorage, Alaska, personal communication). That is to say, the decision which was negotiated met the needs of the several parties and has held up over time.

While the parties to the Terror Lake negotiation were blessed with people skills and worked as policy politicians to some extent, none of the parties used a formal institutional analysis. However, KEA's astuteness in analyzing the negotiation contributed to overall success. The Terror Lake experience illustrates that careful institutional analysis is possible and productive. Such an institutional analysis would include a description of organizational roles and power and could be used to establish negotiation strategies and tactics. Knowledge of organizational roles and negotiation strategies should guide you in selecting the appropriate application of IFIM. Once engaged in the negotiation you should be careful to recognize the resistance of individual negotiators to new ideas and design means to cross those barriers.

Issues Analysis

In a manner similar to the legal and institutional analysis, negotiators will need to identify as many issues of concern from the primary stakeholders as possible during IFIM's initial phase. It is relatively easy to determine how someone's proposed action will affect our own values or those of the agency we work for, so identifying issues that are near and dear to our interests may not be very difficult. Getting all of the issues on the table, not just our own, however, is an important preparatory step in an application

of IFIM. Nothing will derail the resolution of a problem quicker than the alienation of important or potentially powerful stakeholders by ignoring the issues of greatest concern to them. What makes issues analysis so critical is that the derailment often occurs after large amounts of time and money have been invested in a study that is destined to fail for lack of this simple step.

The early planning phase of an IFIM study typically starts with a qualitative assessment of potential impacts associated with some proposed action. Historically, guardian agencies have usually proposed the action (e.g., building a dam, diverting flow, or snagging trees out of a channel), with reaction coming from the advocates. This picture is changing, however, as advocate groups are beginning to initiate the proposed action (e.g., changing the operation of a reservoir), with reaction coming from the guardian side of the role map.

Throughout this text, we frequently refer to impacts, assessments, and evaluations (Fig. 2-6). Impact refers to a human-induced action and its effect (either positive or negative) on selected components of the ecosystem. Assessment involves an analysis of the spatial and temporal deviations from a baseline condition as determined from predicted effects of an alternative proposed action. A baseline is a reference condition, against which comparisons are made. The common feature of baselines, as they are used in IFIM, is that they are intended to represent existing conditions: the current political and institutional climate, present water uses and management, existing channel characteristics, and prevailing thermal and water quality conditions. Evaluation is defined as a process of estimating the values society places on the changes (impacts) to natural resources that will result from a proposed action.

An IFIM study, from beginning to end, is an impact assessment and evaluation of alternatives. This is a very important concept, because many stakeholders hold to the misconception that the purpose of IFIM is to identify a biological minimum flow. An IFIM analysis is much broader than a simple minimum flow determination. In fact, the higher use of IFIM is to assess the effects of minimum flow rules, rather than to determine what the rules should be. (If this distinction is new to you, review incremental and standard setting problems in the primer for IFIM [Stalnaker et al. 1995].) An IFIM study starts with an initial identification of potential impacts and ends with a final evaluation statement. Between these two points lie study planning, implementation, and alternative analyses. This process leads to proposals for problem resolution. Beanlands and Duinker's (1983) view of environmental impact assessment is well-suited to IFIM studies.

> Environmental impact assessment is grounded in the perceptions and values of society which find expression at the political level through administrative procedures of government. Scientists are called upon to explain the relationship between contemplated actions and environmental perceptions and values. (Beanlands and Duinker 1983)

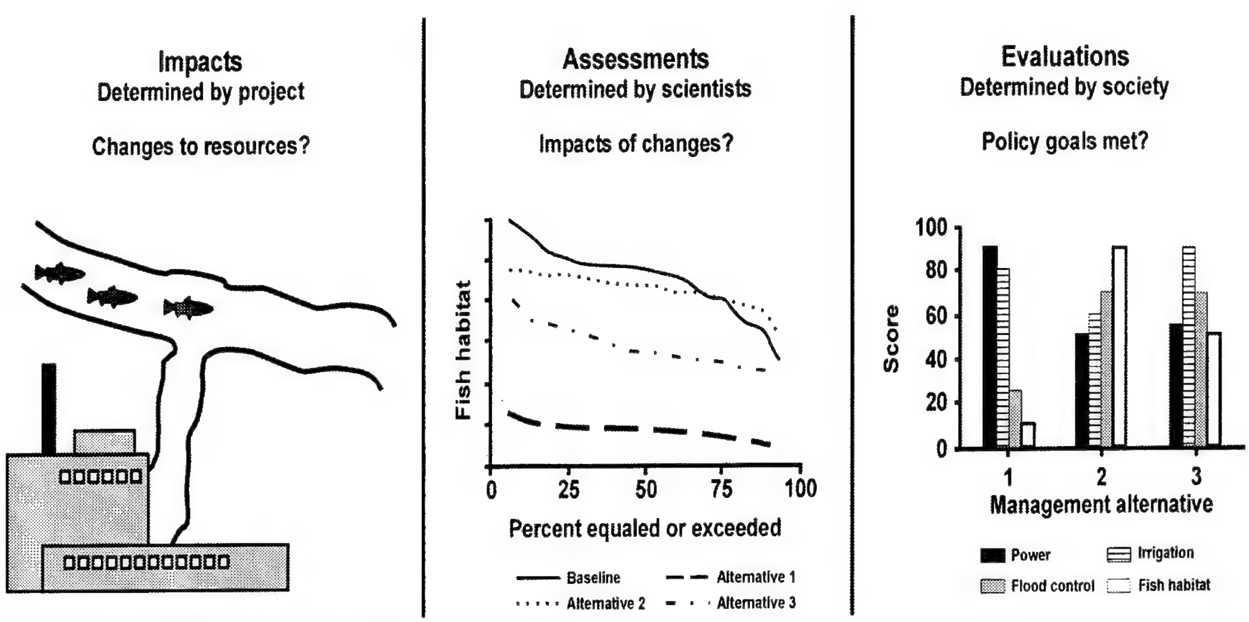

Fig. 2-6. Components of an impact assessment and evaluation study.

Identifying Natural Resource and Human Value Issues

Important natural resources and human values largely determine how the changes resulting from a proposed action will be measured, analyzed, and evaluated. The potential solutions considered are controlled by and restricted to the selected resource and human use activities. Most people view the selection of the valued resources and human activities as an objective decision based entirely on the potential impacts. The decision to address specific resources and values over others, however, is always subjective. Legal mandates, economics, and social, political, institutional, and even personal values determine the resources and activities deemed to be important to a specific evaluation. It is therefore important to get input from all of the parties involved in the decision process.

Not all resources and uses are necessarily equally important, but all values identified by the stakeholders as critical to the decision must be considered. The rationale for selecting or omitting various resource parameters and human use activities should be explicitly stated in the study plan. For example, an endangered species may be assigned a higher priority than a sport fish or a river recreation activity, owing to laws governing endangered species.

Four problems are commonly encountered when trying to identify the important natural resource and human use values for the focus of an instream flow study. First, merely identifying the resource (e.g., sport fish, endangered mussel, or other valued resource) in the study area does not answer the question of where or when impacts related to a proposed action will occur. Second, each identified resource requires an additional commitment of time and effort and adds complexity to the decision process. Third, identifying valued resources for the focus of an IFIM study does not indicate whether a change to the resource is acceptable or not. Such acceptability is a policy decision, not an analytical one. Finally, once a valued resource has been identified, it is frequently assumed that the study will center on population changes. To some extent, by addressing the following questions, you can obtain the information necessary to develop concise study objectives that start the IFIM process in motion:

1) Who has the jurisdiction to identify and speak for the natural resource and human use values?
2) What are their statutory authorities and mandates?
3) What kind of information do they need during the decision-making process?
4) How likely is it that the valued resource will be affected either directly or indirectly by the proposed project?
5) What level of protection is required for each valued resource?

At the start of an IFIM study, participants face the often bewildering task of identifying and organizing potential impacts in a systematic way. There are two especially important aspects of this task that must be addressed during the analysis of issues. First, it is helpful to think comprehensively about the full range of potential impacts arising from the proposed action. A fundamental problem in dealing with multiple issues is simply keeping track of everything without losing sight of an important issue (and alienating a stakeholder). The second consideration is that not all issues will be of equal importance to the resolution of the problem and some may actually get in the way. Thus, it is important to be able to distinguish between a potentially major impact and a relatively trivial factor that will be invisible by the time the study is completed.

Organizing the Issues

The passage of the National Environmental Policy Act of 1969 led to the use of several impact assessment techniques. These techniques are commonly categorized into three types: checklists, matrix tables, and cause and effect diagrams. These techniques are appropriate for the initial identification of potential impacts. The following information for checklists and matrix tables is extracted from Trial et al. (1980) and Westman (1985). The cause and effect technique is summarized from Armour and Williamson (1988).

Checklists are subjective, qualitative, one-dimensional lists of potential impacts for specific actions. The actions are specific to a project type and location. Associated impacts are usually determined by a group of informed individuals and are based on their judgement and prior experience. One variation of a checklist approach is for the evaluators to assign a plus (+) for a positive effect, a minus (-) for a negative effect, or a zero for no effect, to the list of all potential impacts. Positive and negative numerical scoring can be used to give an indication of the severity of each potential impact. Project actions are not quantitatively linked to environmental effects in checklists. The most highly developed checklists use weighting factors that assign a relative importance value to each physical, chemical, or biological impact considered.

Subjectivity is a problem with checklists because they rely solely on the knowledge and experience of the individuals making the judgement. They are, therefore, best done by an interdisciplinary team. Failure to assess specific project impacts is a major criticism of checklists. For example, identifying that an action will impact water quality is not as useful as identifying that dissolved oxygen and turbidity will change. Other criticisms of checklists are that temporal and spatial aspects are not considered, secondary and cumulative effects are not addressed, and scaling techniques and weighting factors give a false sense of precision. Overall, checklists do not assess environmental impacts very accurately, but they are an initial starting point for identifying those issues that are important enough to justify more sophisticated analysis.

Matrices are extensions of the basic checklist. A matrix is used to compare a list of project activities along one axis against a list of physical, chemical, and/or biological parameters along another axis. The project activities and environmental parameters can be general or specific. A simple check can be used to identify potential effects. Leopold et al. (1971) used relative values to indicate both importance and magnitude of the impacts. Like checklists, a multidisciplinary team uses its knowledge and experience to rank the importance of the potential impacts.

Several variations of matrices have been developed. Leopold et al. (1971) developed the early version matrix. Fischer and Davies (1973) used matrices coupled with a three-step process. Improvements over Leopold's matrix are explicit, dealing with short- and long-term impacts and alternative project management, design, and location. The dual matrix (Yorke 1978) was developed as a planning aid to organize information on impacts of water resource development. The method employs two matrices. The first relates water development activities to physical factors (cause → condition) and the second relates these physical factors to biotic components of aquatic ecosystems (condition → impact).

The dual matrix approach overcomes one of the major shortcomings of other matrix approaches by documenting the literature used to make the connection between an activity and affected environmental variables. Matrices tend to be more comprehensive than checklists. By presenting impacts in a two-step process, the technique allows for consideration of short- and long-term and primary and secondary impacts.

Cause and effect diagrams are used to organize potential project-related problems and environmental variables into a framework that is logical, technically defensible, and easy to understand and communicate. Based on a series of concise problem statements, the causes (biological, chemical, physical, social) are linked to effects (environmental changes), which in turn are linked to secondary and tertiary causes and effects (Fig. 2-7). As the chain is constructed, specificity increases with each entry until endpoints are identified (Armour and Williamson 1988). Cause and effect diagrams have been used for cumulative impact assessment (Williamson et al. 1987).

Macrohabitat Issues

Macrohabitat is the set of abiotic conditions such as hydrology, channel morphology, thermal regime, chemical properties, or other characteristics in a segment of river that define suitability for use by organisms. Macrohabitat controls the longitudinal distribution of aquatic organisms. Changes in macrohabitat characteristics are often associated with changes in the composition of communities along various environmental gradients. The gradient in communities has also been termed longitudinal succession when referring to the upstream to downstream sequential changes along a river. The concept of longitudinal succession was developed by Shelford (1911) as he noted that species abundance and composition changed in response to gradational changes in environmental conditions. This concept has evolved into what today is termed the river continuum concept (Vannote et al. 1980).

During the problem identification phase, we must determine potential responses of the four major macrohabitat classes (flow regime, channel structure, thermal regime, and water quality) to an ongoing or proposed change in land or water use. It is far better to identify issues and problems through a logical and rigorous process than to assume

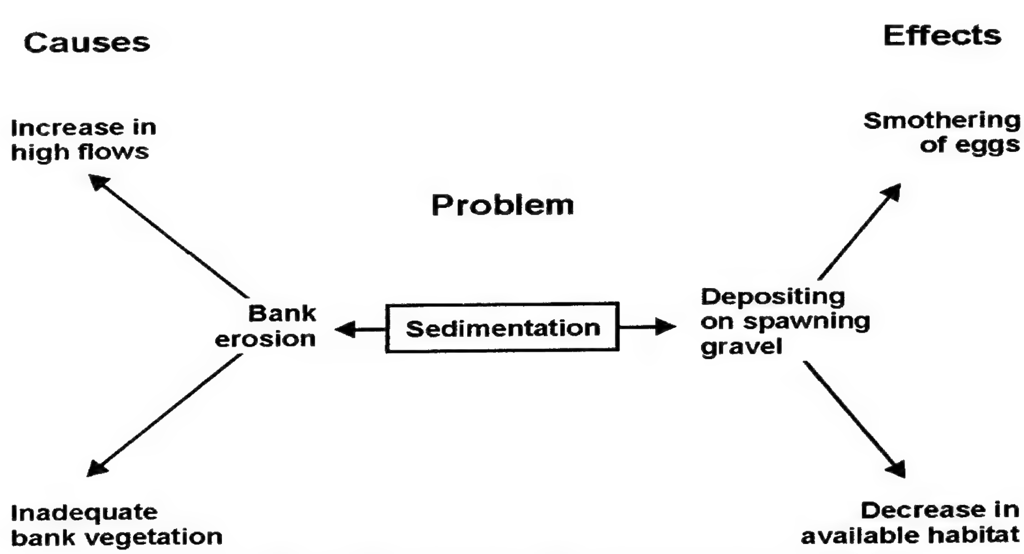

Fig. 2-7. Cause-and-effect diagram used to array potential impacts and issues in an IFIM study.

that they are known *a priori*. The first step is typically to bound the likely problem(s) in space and time. This may be accomplished through a careful study of the proposed project operations, examination of topographic maps of the study area, and participation in a site visit. A preliminary analysis of the issues may involve one or more of the following procedures: (1) data collection to observe and describe the extent of potential changes, (2) use of models to predict changes that might be caused by the proposed project, (3) use of previously gathered information to describe the status and trends of the factors within the proposed project boundaries, (4) joint development of assumptions leading to agreement regarding potentially significant changes, and (5) joint agreement concerning changes judged insignificant enough to be ignored.

Hydrologic Issues

Changes in the hydrologic regime may result from a wide variety of factors, some controllable, some uncontrollable, some intentional, some accidental. Although uncontrollable and accidental changes in the hydrologic regime (e.g., global climate change) are important to riverine resources, intentional changes in water use and management are most often the driving forces behind the initiation of an instream flow study. The following section explores some of the more common hydrologic issues that arise during the problem identification phase of an IFIM analysis: (1) factors affecting the variability of water supplies, (2) quantification of changes in streamflow, and (3) issues related to reservoir operations.

Water budgets. The balance among the various components of the hydrologic cycle are of critical importance in the coexistence of humans and rivers. Scattered amid the inherent uncertainties of annual precipitation patterns are factors that can change the distribution of water among the pathways of the hydrologic cycle. Thornthwaite and Mather (1955, 1957) introduced the concept of a water budget to describe the balance of inflow from precipitation and outflow by evaporation, groundwater storage, and streamflow. The basic concept of the water budget can be summarized as follows:

$$R = P - ET - \Delta SM - \Delta GWS \qquad (1)$$

where R is open channel runoff, P is precipitation, ET is evapotranspiration, ΔSM is the change in soil moisture, and ΔGWS is the change in groundwater storage. All units in equation 1 are measures of length (inches, feet, meters) and can be converted to volume units by multiplying all terms by the area of the watershed above the location where runoff (R) is measured. To convert R to a unit of discharge, the volume units are divided by a time interval (e.g., the number of seconds in a month to determine mean monthly discharge in cubic feet or cubic meters per second).

Historically, water budgets of this type have not been used extensively in instream flow studies for a number of reasons. The most practical reason has been that projects involving deliberate modifications to the hydrologic regime (e.g., diversions, reservoir operations) have been given higher priority. A second reason is that constructing a water budget model for a network of streams is not easy and its accuracy depends on the modeler's ability to estimate rate parameters for evapotranspiration, soil moisture, and groundwater storage. As instream flow studies shift from the narrow issues of water management to the broader spectrum of ecosystem management, water budgets will probably be used more often.

Quantifying hydrologic changes. When an intentional change to the hydrologic regime is proposed, several issues emerge during the problem identification phase of IFIM: (1) What is the magnitude of the change? (2) Is the change measurable? (3) How often will the change occur? (4) When will the change occur? (5) Is the change sufficient to warrant further analysis? Two tools are particularly valuable in evaluating these issues, the hydrologic time series and the flow duration curve. Both techniques are useful for comparing the baseline hydrologic regime with the hydrologic regime that would result with the proposed alternative in place.

A hydrologic time series is a chronological distribution of streamflow at a particular location. When examining a hydrologic time series, it is important to understand what kind of discharge-related statistics are being portrayed in the series. For example, the discharges portrayed in Fig. 2-8 are base flows (streamflow that occurs in the absence of significant precipitation or runoff, contributed solely from groundwater). Fig. 2-8 could have depicted such things as the instantaneous annual peak discharge or average annual, monthly, weekly, daily, or hourly discharge just as easily. Make sure you know what you are looking at when you examine a hydrologic time series.

One way to express an impact, or to decide if it is important enough to elevate to the status of an issue, is to assess the frequency with which things happen in the time

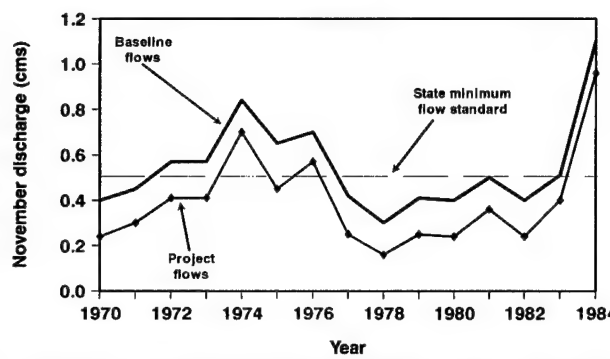

Fig. 2-8. Hydrologic time-series comparing average November streamflows under baseline conditions with those with a hypothetical diversion in place. The horizontal line represents the State's recommended minimum flow standard.

series. For example, we could decide whether a proposed project will substantially affect the flow regime by determining how often a state's minimum flow standard would be violated with and without the project in place. The hydrologic time series is handy for addressing this type of question. Examination of Fig. 2-8 shows that the minimum flow standard was violated eight times under the baseline. With the project in place, it was violated 12 times, or an increase in the frequency of violations of nearly 50%. In most jurisdictions, a 50% increase in violations would probably be considered an issue.

Hydrologic time series data can be used to quantify changes in the overall availability of streamflow, using common statistics such as the sum, mean, and standard deviation. In the example illustrated in Fig. 2-8, the average November discharge under the baseline condition was 0.55 m^3/s (cms), with a standard deviation of 0.20 cms. With the diversion in place, the average November discharge would have been 0.41 cms (standard deviation of 0.20 cms). By using equation 2, we find that the difference between the two average flows is 0.14 cms, or -25.7%.

$$\Delta Q_{base} = \frac{(Q_{proj} - Q_{base})}{Q_{base}} = -25.7\% \qquad (2)$$

where Q_{base} is the average discharge under the baseline condition (0.55) and Q_{proj} is the average discharge with the project in operation (0.41). Interestingly, while the project will result in a 25% reduction in the mean, the year-to-year variance in discharge is unaffected, as reflected by the standard deviation.

Hydrologic time series data are available in tabular format more often than graphically. It is quite simple to import the tabular data into a commercially available spreadsheet and obtain such statistics as the mean, standard deviation, minimum, and maximum values. Suppose, however, that we are more interested in a few events, such as drought years, than we are in the entire time series. For example, the project depicted in Fig. 2-8 looks like it might have a greater impact in 1978 than in 1974. To answer questions such as these, we need to analyze frequency and magnitude of flow events at the same time by using what is known as a flow duration curve (Fig. 2-9).

A flow duration curve is a plot of a discharge statistic (e.g., mean November streamflow) versus its cumulative empirical probability of occurrence in the hydrologic time series. The curve is derived from a duration table (Table 2-2), wherein the discharge statistics are arranged in descending order rather than chronological order. Each discharge in the table is assigned a rank from 1 (highest flow) to n (lowest flow), and its cumulative probability (P; plotting point in Table 2-2) is calculated by:

$$P = \frac{m}{n+1} \qquad (3)$$

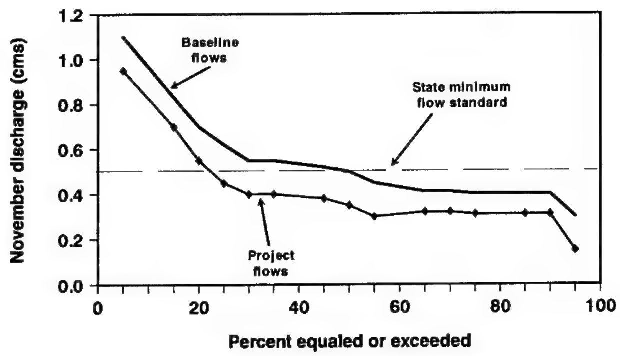

Fig. 2-9. Flow duration curves comparing average November streamflows under baseline conditions with those with a hypothetical diversion in place. The horizontal line represents the State's recommended minimum flow standard.

Table 2-2. Example of a flow duration table for the hydrologic time series shown in Figure 2-8.

Year	Historic discharge (cms)	Discharge with project (cms)	Rank (*m*)	Plotting point (*m*/*n* + 1)
1984	1.105	0.96	1	0.0625
1974	0.85	0.71	2	0.1250
1976	0.71	0.57	3	0.1875
1975	0.62	0.48	4	0.2500
1973	0.57	0.42	5	0.3125
1972	0.57	0.42	6	0.3750
1983	0.54	0.40	7	0.4375
1981	0.51	0.37	8	0.5000
1971	0.45	0.31	9	0.5625
1977	0.42	0.28	10	0.6250
1979	0.42	0.28	11	0.6875
1982	0.40	0.25	12	0.7500
1980	0.40	0.25	13	0.8125
1970	0.41	0.25	14	0.8750
1978	0.31	0.17	15	0.9375

where m is the rank and n is the total number of events in the time series. When formulated in this manner, the plotting position represents the exceedance probability, or the probability that the associated event will be equaled or exceeded. (Note: Had the discharges been ordered from low to high, the plotting position is the probability that the associated event will not be exceeded.) In Fig. 2-9, the exceedance probability represents the probability that a particular flow or a greater flow will be present during November (e.g., there will be at least x amount of water y percent of the time). When equation 3 is inverted ($(n+1)/m$), the result is known as the recurrence interval or return period, defined as the average time interval between events equalling or exceeding a given magnitude. Recurrence intervals are commonly based on annual flow data and reported in years (e.g., the 100-year flood event).

Going back to the example from Fig. 2-8, suppose that our objective is to quantify the change in discharges only for the drought years and that we define a drought as the flow that is equaled or exceeded 75% of the time. Table 2-2 shows these flows occurred during 1970, 1978, 1980, and 1982. The average of the baseline discharges for these years was 0.38 cms, and the average with the project in operation was 0.23 cms, or a difference of -37.7%. A 10% deviation from the baseline condition is often used as the criterion for making an issue out of hydrologic changes. However, the streamflow statistic used to make this decision can be quite variable. Some practitioners use the mean annual flow for this decision, whereas others prefer a more conservative measure, such as annual base flows or the average of several drought years.

The flow duration curve can also be used to interpret the hydrologic character of a stream or changes thereof. Fig. 2-10 shows three examples of flow variability as related to a flow duration curve. A horizontal line means that there is virtually no variability in the system over time. As flow variation increases, the angle of the flow duration curve deviates more from horizontal, with a vertical line representing chaos. A bimodal flow distribution, common below hydropeaking facilities, appears as a near-vertical line with a horizontal "wing" at each end. The duration curve can also be used to interpret changes in a flow regime resulting from the implementation of an alternative. For example, if the curve for a proposed alternative lies entirely beneath the baseline curve, the streamflow will always be lower with the project in operation than it was under the baseline. When flow duration curves cross each other, it means that one of the regimes is more variable than the other (e.g., the high flows will be higher and the low flows lower). If the alternative flow regime is less variable than the baseline, there is probably a storage reservoir in the works.

Key Points About Hydrologic Time Series and Duration Curves

Hydrologic time series and duration curves contain the same basic data but are displayed differently. Pertinent differences between the two types of displays are contrasted below:

Complexity - A large amount of data can be conveniently displayed on a single flow duration curve without loss of resolution. Time series data become more difficult to consolidate, display, and interpret as the number of events in the series increases.

Chronology - The sequence of events is preserved in a time series display but obliterated in a duration curve.

Quantification - Determining total differences between two hydrologic time series is equally easy using either the time series or the duration curve approach. Quantification of differences between specific events (e.g., during a 1-in-4 year drought) can be done best from a duration curve.

Format - Time series plots are consistent in that time is always expressed on the x-axis and discharge on the y-axis. Duration plots can come in a bewildering array of formats and construction techniques (Fig. 2-11). Sometimes the scales are reversed or arrayed in ascending order, so that the probabilities are for nonexceedance rather than exceedance. The scales can be arithmetic, logarithmic, semi-logarithmic, or probit (a logarithmic scale wherein the spacing decreases from probabilities of 0-50% and then increases from 50-100%). Because individuals become accustomed to seeing duration plots in a particular format, they may find it difficult to interpret curves presented differently.

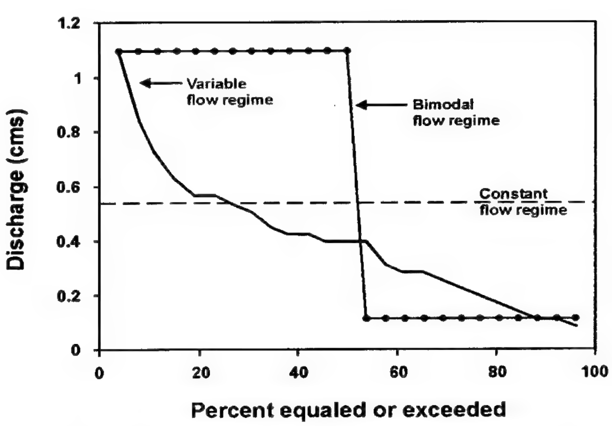

Fig. 2-10. Examples of different flow duration curves representing hydrologic regimes ranging from constant discharge to high temporal variability.

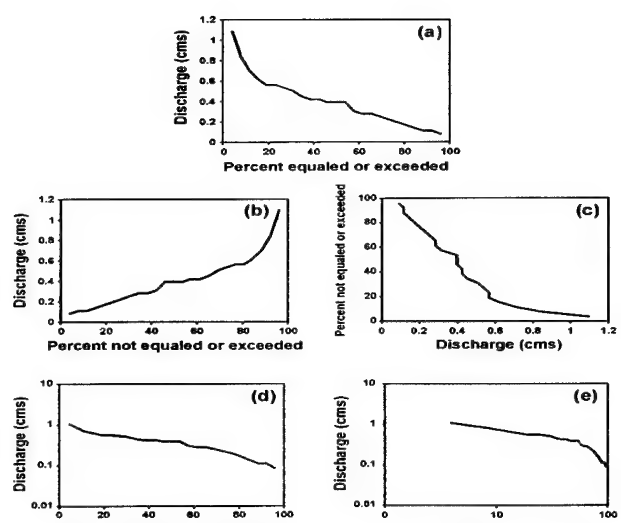

Fig. 2-11. Different formats in which flow duration curves may be displayed, using the same original streamflow data: (a) arithmetic plot, probability on x-axis, (b) arithmetic plot, sorted in ascending order (notice x-axis label), (c) x and y axes reversed, (d) semi-log plot, and (e) log-log plot.

Reservoir issues. It is probably fair to say that most instream flow studies involving IFIM also involve a reservoir. In part, this phenomenon is due to the large number of impounded rivers in the United States. It is almost impossible to find a moderate-sized free-flowing river outside of Alaska. Furthermore, because of the number of conflicting interests in reservoir operations, problems associated with reservoirs are nearly always incremental (Stalnaker et al. 1995). Sometimes, a study will be designed to evaluate alternative operations of an existing project. Sometimes, the goal is to minimize or mitigate the environmental impacts associated with construction and operation of a new reservoir.

Reservoir operators often partition storage according to the model illustrated in Fig. 2-12. A portion of the storage capacity, called dead storage, is reserved for sediment accumulation between the bottom of the reservoir and the outlet. Most multipurpose storage reservoirs also have some freeboard in the upper levels reserved for flood wave attenuation. The active storage of a reservoir exists between the top of the dead storage level and the bottom of the flood storage level. Priorities often exist among various uses of reservoir storage. Flood retention usually has a very high priority in reservoir operations, even if the reservoir was not constructed primarily for flood control. Active storage may be parceled out for irrigation, municipal water supplies, production of electricity, instream flow requirements, downstream recreation, and re-regulation of flows from upstream reservoirs. In addition, most reservoirs provide valuable flat-water recreation and aesthetics. Conflicts are commonplace between users of the reservoir (who prefer the reservoir to be full to the brim all the time) and users of the released water (who will draw the reservoir down to dead storage if necessary to meet their needs). In addition, conflicts may exist among users of the reservoir pool (e.g., marina owners vs. managers of wildlife refuges) and among users of the downstream release (e.g., irrigators vs. producers of electricity).

One of the most interesting issues related to reservoir operations is the existence of a built-in feedback loop. That is, one cannot simply assess the impact of reservoir storage on instream flow releases. Assessing the impact of instream flow (and other) releases on reservoir storage is also necessary. For example, suppose a high instream flow release were implemented without consideration of its impact on reservoir storage. If the release is high enough and lasts long enough, the active storage of the reservoir could be exhausted. Even if the reservoir pool stakeholders did not complain, the resulting instream flow would equal the inflow to the reservoir because there would be no storage to draw on. At some time, the instream flow release will have to be less than inflow in order to fill the reservoir back up.

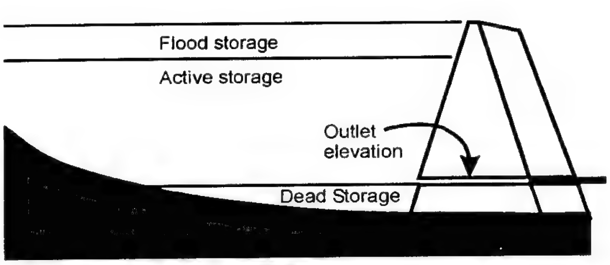

Fig. 2-12. Allocation of reservoir storage capacity according to purpose.

Channel Dynamics and Stability

The structure, pattern, and dimensions of the river channel interact with discharge to control or influence the availability of instream habitat at several scales. For example, a narrow stream will be more influenced by shading than a wide stream and may not get as warm during summer. A stream with a simple riffle-pool pattern may not offer as many types of meso- or microhabitats as a braided channel. Generally speaking, issues associated with channel dynamics and stability are of two types: determining flow requirements to prevent the channel from changing and predicting how channel changes are likely to affect instream habitat. The potential importance of channel change in an IFIM study depends on the type of channel and the type of perturbation.

Channel changes result from erosion and deposition, and some types of channels are more dynamic than others. A bedrock controlled stream, predominantly incised in solid rock, may be highly resistant to erosion but may change over time through depositional processes. Colluvial streams are littered with material deposited in a channel by avalanches, landslides, glaciers, or catastrophic floods. A colluvial stream may be somewhat more vulnerable to erosion than a bedrock channel, but it is probably also exposed to landslides or other depositional events on a regular basis. The most dynamic kind of channel is alluvial, being formed by simultaneous processes of erosion and deposition of sedimentary materials. Alluvial channels are self-adjusting. If the balance between streamflow and sediment production from the watershed is changed, the channel will adjust to conform to the new set of conditions (Dunne and Leopold 1978). There is an approximate balance between the amount of sediment supplied to the stream and the stream's ability to transport sediment under the prevailing flow regime. When this balance is consistent over time, the channel is said to be in a state of dynamic equilibrium. An equilibrated channel is not a static channel. Riffles may be scoured into pools, pools may become riffles, meanders will migrate, bars will form and disappear. Despite the instability that might be evident at a single location, however, there is a remarkable consistency in channel pattern and cross-sectional profile for the entire stream. The proportion of riffles and pools, meander

wavelength, sinuosity, and width-to-depth ratio all remain relatively constant. Even a braided channel, which is defined by a constantly shifting streambed, may represent an equilibrium condition (Leopold et al. 1964).

During the problem identification phase of an IFIM study, one of the issues to be resolved is whether or not a proposed action will lead to an episode of channel disequilibrium. If such disequilibrium is anticipated, the discussion quickly shifts to the nature of the change, what can be done to prevent it, and if change is imminent, how to address its impacts. Generally speaking, channel changes can be classified into three broad categories: (1) channel enlargements or reductions, (2) channel aggradation or degradation, and (3) changes affecting the size distribution of channel materials.

Channel enlargements and reductions. Changes in water use or land use that increase runoff often result in an increase in the magnitude and frequency of high flow events. Increased frequency of channel-forming discharges (known as dominant or effective discharges) are often associated with increased bank erosion. Consequently, one of the first indications of channel enlargement is an increase in width. During channel enlargements, the width-to-depth ratio may stay about the same, so an increase in depth may occur at the same time. Because meander wavelength and riffle spacing are both related to channel width, an increase in channel width may also signal an impending reduction in sinuosity and a lengthening of the distance between riffles.

Predictably, channel reductions result from conditions that are diametrically opposed to those causing enlargements, but with the added factor of vegetational encroachment. When the dominant discharge is reduced, areas near the stream margins or on sand bars can become colonized by vegetation. As the vegetation takes hold, it not only stabilizes the deposit but acts as a sediment trap when the deposit is inundated. Over time, the deposit accretes in elevation and will ultimately form a natural levee or a new bank.

Aggradation and degradation. Aggradation and degradation are responses of the channel to an imbalance between sediment inflow and the transport capacity of the stream. Aggradation occurs when the supply of sediment exceeds the transport capacity of the stream. Aggradation can occur if sediment production from the watershed increases without a corresponding increase in dominant discharge. This form of disequilibrium can also occur if sediment production remains constant and the dominant discharge is reduced either in frequency or magnitude. The imbalance between sediment supply and transport capacity results in deposition and storage of sediment in the active channel. If the cause of the aggradation is a reduction in streamflow with no change in sediment yield, the result will likely be bank-building and channel reduction as described earlier. If aggradation is caused by an increase in sediment production with no change in discharge, the most typical channel response is an increase in the width-to-depth ratio. Sediment is stored in the channel, which decreases the depth and increases the erosive forces against the banks, which widens the stream. In extreme cases, aggradation can lead to a wholesale change in channel pattern and structure from straight or meandering to braiding.

Degradation results when the transport capacity of the stream exceeds the sediment yield from the watershed. The construction of dams in alluvial streams is probably the most common cause of channel degradation. The sediment normally carried by the stream is deposited in the reservoir. The energy once dissipated by moving the sediment in the channel is now available to the stream to erode its bed. As the channel bottom is eroded downward, the shallow portions of the streambed are inundated less frequently. As the stream continues to degrade, the old streambed becomes the new floodplain and the old banks become terraces. The width-to-depth ratio of the incised channel may be considerably less than that of the original. Consequently, when the degraded channel finally equilibrates, it will commonly have a sinuosity much greater than the original channel.

Channel materials. Changes in channel pattern and structure are commonly accompanied by alterations in bed material composition. Modification of the substrate is almost a foregone conclusion when aggradation or degradation occurs in an alluvial channel, but substrate changes can occur in bedrock and colluvial channels as well. Although these channels might be less susceptible to channel changes than alluvial channels, they may be vulnerable to more subtle, but equally detrimental, changes in substrate.

Essentially, there are only two types of substrate changes: those that cause the particle size distribution to become coarser and those that cause it to become finer. The particle size distribution refers to the mixture of different-sized materials in the substrate matrix. Embeddedness is a subset of the particle size distribution, referring to the degree to which the interstitial spaces between the larger substrate materials are filled with fine particles (generally silts and sands). Both aspects of particle size distribution can have important biological ramifications.

When a stream degrades, the smaller materials in the substrate matrix are more easily eroded than the larger materials. Sands and silts are removed first, leaving behind the gravel, cobbles, and boulders. Then the gravels are eroded, leaving the cobbles and boulders. Eventually, the boulders may be the only materials left in the channel that cannot be moved by the stream. This process of continually winnowing away the smaller materials and leaving a veneer of larger ones is known as armoring.

Although armoring is a common result of channel degradation in alluvial streams, a related phenomenon can occur in bedrock or colluvial channels. Alluvial materials, such as cobbles and gravel, are normally recruited to bedrock channels during high runoff events. As the discharge recedes, some of these particles will be deposited in the channel. During the next high flow episode, the deposits are eroded and then replaced. If the source of these materials is suddenly cut off (e.g., by the construction of a dam), previously deposited alluvium will be removed but will not be replaced. In a very short time, there will be substantially fewer, less extensive deposits of gravel and cobble in the channel, if there are any left at all.

A stream undergoing channel reduction or aggradation will often experience a decrease in particle size and increased embeddedness. As there is more sediment load in relation to the sediment transport capability, the stream has less energy available to move all sediment sizes. Eventually, a veneer of silt and sand can be deposited over the previous streambed surface. Sometimes, the depth of the deposit can be more than a thin layer. Leopold et al. (1964) reported that the surface elevation of the Rio Grande increased by nearly 4 m between 1895 and 1935.

Key Points About Channel Change

The three basic types of river channels are bedrock, colluvial, and alluvial. Bedrock channels are incised in solid rock strata. Colluvial channels contain materials deposited by mass wasting of valley walls, avalanches, and glacial deposits. Alluvial channels are formed in materials that were previously deposited by the stream under the recent hydrologic regime.

Alluvial channels are more active and more prone to change than either bedrock or colluvial channels. The channel is in a state of dynamic equilibrium when the sediment load is in balance with the stream's transport capacity. When the load exceeds transport capacity, sediment will be stored in the channel, either through channel reduction or aggradation. When the transport capacity exceeds the load, the channel will readjust through channel enlargement or degradation.

Episodes of channel disequilibrium in alluvial channels may result in wholesale changes in channel pattern and structure. Bedrock and colluvial channels are relatively immune from erosive forms of disequilibrium, but both can be affected by channel-filling events.

Channel armoring results from the progressive winnowing of fine materials from the substrate during channel enlargement and degradation. A phenomenon similar to armoring can occur in bedrock and colluvial channels when the supply of alluvial materials is cut off from the channel. The amount of fine materials incorporated in the substrate (embeddedness) generally increases during disequilibria involving channel filling.

Temperature and Water Quality

Water temperature is affected by streamflow in several ways: (1) streamflow determines the volume of water that must be heated or cooled, (2) stream width influences exposure to heat sources, and (3) velocity influences the time of exposure to heat. Though general tabulations of lethal temperatures are available (U.S. Environmental Protection Agency 1986), especially for commercially valuable species, we recommend searching for literature specific to the target organisms. The surveys can be valuable starting points for your search. Unfortunately, laboratory studies may provide unrealistic thermal values. As discussed by Bartholow (1991), field studies provide the most useful benchmarks as they are a better indicator of the exclusionary nature of sublethal water temperature than are the strictly lethal limits. This principle is especially true since reproductive activities tend to be the most restrictive, yet remain little studied (Brett 1956). Two other notes about water temperature are worth considering during problem identification. First, nonlethal daily thermal fluctuations can also influence community structure (U.S. Environmental Protection Agency 1986). Unfortunately, no one is able to make predictions, as little is understood about the importance of these diel fluctuations. Second, it is common to apply a 2° C safety factor to protect biotic communities from elevated temperatures when the limits you apply originated from laboratory studies (Coutant 1976).

Since water temperature is so often an important variable, it may be useful to outline the set of thermal effects a bit more thoroughly. Thermal effects may influence fish populations through (1) directive factors, (2) controlling factors, (3) lethal factors, (4) growth factors, and (5) synergistic factors (Fry 1947). Directive factors influence the timing of fish behavior (sometimes called biological periodicity). Water temperatures can trigger movement within the system, based on the thermal gradient, and influence the initiation of spawning behavior. (Note: if target species are migratory, secondary areas may be impacted. If so, then broadening the study area may be required.) Controlling factors govern process rates and act to determine the duration of the periods from incubation to hatch and from hatch to emergence. Thermal units called degree-days are useful to measure controlling factors. Lethal factors and fish growth factors are often considered a direct function of a species' thermal unit experience, assuming that the experience directly affects metabolic regulation. Growth is limited to a relatively narrow temperature range, usually having a thermal optimum with declining growth rate, or even growth inhibition, at temperatures on either side of the optimum. The relationship between growth and temperature, however, is complicated by a host of intervening factors that may either shift the optimum or influence the shape of the functional relationship. Synergistic factors are simply those that influence the biological response to other potential limiting factors. For example, water temperature directly affects how an organism reacts to waterborne toxins.

The essence of making judgements about possible future system states relies on understanding the present

conditions. The current situation can also provide clues on problems or borderline conditions that need further study. Occasionally, water temperature or quality problems are self-evident in the form of fish kills. Determining the exact cause can be tricky, but the procedures are thoroughly described in Meyer and Barclay (1990). More typically, acquiring existing water quality data would usually be the first choice in most macrohabitat studies. Often a quick perusal of the existing data will tell you whether, or how often, extremes have been reached. Occasionally, this information may be all you need. However, existing data typically may not allow you to construct any relationship between flow and water quality. More important, existing data *will not* be sufficient to describe what will take place if the system is changed in some way. Again, your conceptual models must be integrated here.

Water quality data are collected by many State and Federal agencies (Table 2-3). A publication by the U.S. Geological Survey (USGS; Pauszek 1972) attempted to tabulate all the agencies involved. Though not up-to-date, this publication tabulates the agencies by state and partitions the collections by lakes, reservoirs, canals, estuaries, drains, springs, and wells. In addition, the frequency of measurements is also tabulated. Although the publication does not present any data, it does contain a 194-reference bibliography of sources that do. Out of 7,500 stations currently collecting surface water quality data, about 4,500 are Federal and 3,000 are State. Most (over 4,000) are east of the Mississippi, and most of these are in the Great Lakes states. Of the stations in the West, most measurements are taken in the three coastal states. The number of stations is not so impressive when frequency of measurement is considered. Continuous water quality measurements are made at only 731 stations, largely concentrated in Washington, Oregon, and California. Even if data are from grab samples, however, you may be able to discern the information you need. For example, if you see a dissolved oxygen (DO) reading of 14 mg/L at 3:00 p.m. and the temperature is 27° C, you can safely assume that the DO will be around 2-3 mg/L at 4:00 a.m. due to algal oxygen consumption at night.

If you need additional assistance in identifying data sources or interpreting the data that you find, you may consult a state-by-state directory of water quality contacts (Conservation Technology Information Center 1993). A variety of streamflow, water temperature and water quality data are available on commercial CD-ROMs. These products are compiled from USGS data sets on daily mean flow, daily peak flows, daily water temperature, and daily water quality, as well as from the U.S. Environmental Protection Agency's STORET data base, and Environment Canada's HYDAT. For users with access to the Internet, daily flow and peak flow records are available for most states on the USGS Home Page on the World Wide Web (www.usgs.gov).

Forecasting Macrohabitat Changes

So, how do you proceed in the problem identification phase? First, carefully examine the description of the project or proposed alternative. Second, array the list of known first-order macrohabitat effects. These are likely to include (1) changes in the amount and/or timing of flow, such as volumetric changes (e.g., consumptive use changes), hydrograph modifications (e.g., water management/reservoir operations), or pulsing operation; (2) stream habitat modifications such as inundation or channelization; and (3) changes in loading rates or initial conditions for temperature or water quality constituents such as nutrients or other organic material. Third, compare the magnitude of first-order hydrologic effects during a chosen baseline period for the proposed alteration with the baseline condition for different water supply conditions (e.g., normal water years, wet years, drought years). And finally, estimate the effects that the proposed changes may have on macrohabitat suitability.

Let us look at some examples. For an irrigation project or other consumptive use situation, compare mean monthly

Table 2-3. State and Federal agencies that collect water quality data.

State agencies	Federal agencies
Agricultural departments	Bureau of Land Management
Game and Fish departments	Bureau of Mines
Geologic surveys	Bureau of Reclamation
Pollution control departments	Federal Energy Regulatory Commission
Public health departments	International Boundary and Water Commission
Sanitary engineering departments	National Marine Fisheries Service
Utilities (also local)	Naval Facilities Engineering Command
Water districts	Natural Resources Conservation Service
Water quality control departments	Tennessee Valley Authority
Water resources departments	U.S. Army Corps of Engineers
	U.S. Agricultural Research Service
	U.S. Atomic Energy Commission
	U.S. Environmental Protection Agency
	U.S. Fish and Wildlife Service
	U.S. Forest Service
	U.S. Geological Survey

flows for the consumptive use period. Other water management problems may require comparing mean monthly flows for the entire year. For hydropeaking situations, you may sample weekly operations for specific parts of the year and compare daily maximum and minimum flows with baseline monthly maxima and minima. For stream habitat modifications, calculate the kilometers of stream inundated or channelized. For thermal or wasteload allocation situations, identify point sources and obtain baseline data for appropriate variables for critical periods. For example, if water temperature is a problem, June, July, August, and September may need to be examined, especially in drought years. If dissolved oxygen is likely to be a problem, you may need to examine both summer and winter conditions, especially in hot, dry years. If ammonia is likely to be a problem, winter conditions are likely to present the worst case situation during drought years. If pH is the target, the base flow period is probably the most critical. If pesticides are the likely problem, wet summers may be all you need to consider. Often, an elementary "red flag" analysis of potential problems using a simple model or consultation with experts may be all you need to determine the potential for trouble. If existing baseline conditions are already marginal, temperature and/or water quality monitoring should always be incorporated in the study plan, as these conditions are likely to degrade.

Of course, change per se is no cause for immediate alarm. It is incumbent on the analyst to understand enough about the target organisms to estimate whether the changes are likely to be significant. As an example, water temperature has many potential effects on fish. Temperature may affect net biomass gain, growth rates, consumption rates, digestion rates, survival, gill ventilation, body temperature, metabolism, physiology, respiration, stress level, ionic regulation, energy level, energetic response, behavioral responses, activity, movement, locomotion, ecology, distribution, competition, predator/prey relationships, parasites and disease, migration, reproduction, egg incubation, egg development, and/or synergistic relationships. Worrying about each one, however, is not likely to be fruitful. Instead, developing a list of all species (fish, macroinvertebrates, etc.) *and* life stages which use the "potential" impact area and noting when they are present during the year, where they are, and how they are using the stream may be helpful. You may also want to gather as much biological information as possible from the literature, State fish and game reports, private consulting firms, or universities on the tolerances of the target species.

As with much of the work implied by an IFIM plan of study, one should carefully ask who will be responsible for performing a water quality problem identification. Do not overlook the possibility that a team of people with a wide variety of skills should be involved in problem identification. Always include information producers and consumers on your scoping team. Though scoping potential problems should remain a relatively simple task, more sophisticated analyses require a more thorough knowledge and commitment, and such analyses are performed in the third phase of IFIM.

In many circumstances, models may be the only means of quantitatively describing the cumulative effects of proposed changes on water quality and temperature. But at this stage of the problem identification game, we are more interested in bounding or scoping the problem, not in accurate quantification. You will probably need to rely more on mental or conceptual models to determine what needs further investigation through a formal plan of study. The factors mentioned above can be mentally evaluated by examining the land and water uses in the watershed and their associated primary and secondary effects on measurable variables.

As you can see from the generalized impact matrix (Fig. 2-13), second-order effects may be biologically more important than first-order effects. Data sources and the evaluation process are similar to analysis of first-order effects, but revealing second-order effects without the use of modeling is difficult. Sometimes quite sophisticated modeling may be required to evaluate second-order effects. During problem identification, however, you must rely on your own mental models, from crude to sophisticated, as a guide for the scope of the study plan. In the absence of modeling, analogous situations may prove helpful in determining effects and probable outcomes. The best advice to follow, however, is when in doubt about a possible effect, include it in the study plan.

Microhabitat Issues

Microhabitat and macrohabitat combine to create the total habitat available for organisms. Macrohabitat controls the general pattern of species distribution and abundance, governs the flow of energy through the system, and controls the distribution and abundance of microhabitat

Affected habitat	Impoundment	Channelization	Grazing	Logging	Surface mining	Row crop agriculture	Groundwater mining	Flow diversion	Flow augmentation	Urbanization	Irrigation	Hydropeaking
Sediment yield	X		X	X	X	X				X	X	O
Water yield	O		X	X	X	X	X	X	X	X	X	
Channel morphology	X	X	X	X	X	X			O	X	X	O
Substrate characteristics	X	X	X	X	X	X			O	X	X	X
Cover	O	X	X	X	X	X				X	X	O
Timing of flows	X						O	O	O	O	X	X
Magnitude of peak flows	X	O	X	X	X	X				X		X
Magnitude of low flows	X		O	O	O	O	X	X	X	X	X	X
Thermal regime	X		O	O	O	X	O	X	O	X	X	
Water quality	O		X	X	X	X		O	O	X	X	

Fig. 2-13. Generalized environmental changes associated with land and water uses. X indicates first-order impacts, O indicates secondary effects. From Bovee (1982).

features. Microhabitat availability may affect populations directly through acute survival mechanisms (e.g., availability of refugia for fry during floods) or indirectly through density-related growth and condition factors (e.g., long-term availability of feeding stations, adult growth and condition, adult overwinter survival rates).

Microhabitat is defined by spatial attributes (e.g., depth, mean column velocity, cover type, and substrate) of physical locations occupied or used by a life stage of a target species sometime during its life cycle. In most applications, the hydraulic variables of depth and mean column velocity and the structural variables of cover type and substrate are used to quantify microhabitat in IFIM. Other physical variables can be added or substituted, provided that they are either hydraulic or structural in nature. For example, nose velocities (e.g., measured a few centimeters off the streambed) can be substituted for mean column velocities, if desired, to obtain a more realistic depiction of microhabitat. Substrate may be used as a structural variable for one life stage of a species (spawning microhabitat) but may be replaced by cover for another (adult feeding stations). For some life stages, cover and substrate will be equally important, so methods for incorporating both variables must be devised.

Several microhabitat-related issues may surface during the initial phases of planning an IFIM study: (1) selection of appropriate target species, (2) determination of critical life stages and microhabitat types, (3) alleviation of known or suspected habitat bottlenecks, (4) knowledge of the habitat requirements of a species, (5) temporal variations in habitat use, and (6) spatial continuity, interspersion, and fragmentation of microhabitat.

Selection of Target Species

The selection of one or more target species is a necessary step in an IFIM application, so it might seem strange that the choice of target species could be a source of contention. The target species can become controversial because the choice may be interpreted as a statement of policy. Different species occupy different micro- and mesohabitats. Not all mesohabitats nor all species react the same way to changes in discharge. Some mesohabitats are more susceptible to dewatering than others, for example. When species that occupy these mesohabitats are chosen as the target species (especially if they are chosen exclusively), a policy of protecting or enhancing low flows is implied.

Sometimes, the selection of a particular target species can be interpreted as an attempt to manipulate the results of the analysis. For example, consider these two choices for a target species in a warmwater stream in Alabama: the largemouth bass (*Micropterus salmoides*) and the greenbreast darter (*Etheostoma jordani*). The bass is a microhabitat generalist, but tends to avoid areas in the stream having noticeable velocities. The darter is a microhabitat specialist, occurring almost exclusively in swift riffles over coarse substrates. Because of the differences in the mesohabitats occupied by these species, their preferred microhabitats respond in opposite fashion to changes in discharge (Fig. 2-14).

Largemouth bass have an affinity for low velocities, so bass microhabitat will be greatest at a near-zero discharge and will decrease rapidly as the flow increases. In contrast, the mesohabitats inhabited by the darter will be high and dry at low flow and will not achieve their maximum areas until a substantially higher discharge is reached. Consequently, a group of stakeholders wanting to justify the smallest possible release from a reservoir will insist on the bass as the target species. Stakeholders trying to justify a relatively high minimum flow will favor the darter.

The most obvious answer in selecting target species is to select a mix of species representing a variety of meso- and microhabitats. However, this solution is not perfect either, because the management alternatives favoring one species may work against another. This dilemma has lead some researchers to recommend habitat-use guilds (groups of animals that all use similar meso- and microhabitats). In reality, however, the selection of a particular species or group of species implies the selection and management of particular microhabitats, no matter how deeply the implication is hidden. Although we recommend the use of a mix of target species, be aware that microhabitat implications exist with any selection. Selecting more species may facilitate getting the study started but will ultimately make the analysis of alternatives more difficult.

Critical Microhabitats, Life Stages, and Habitat Bottlenecks

These three issues are so intertwined that it is almost impossible to talk about one of them without discussing

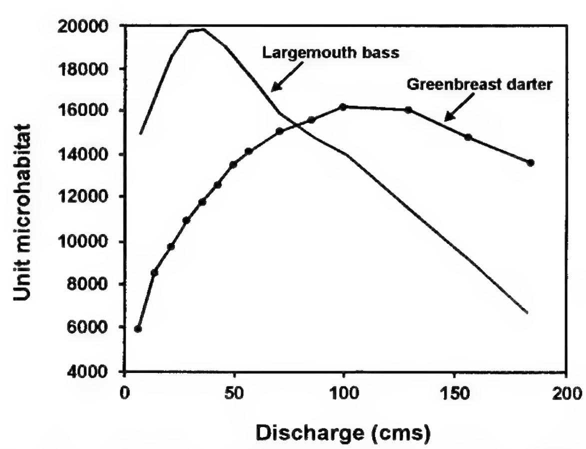

Fig. 2-14. Comparison of typical functional relationships between discharge and microhabitat area for a lacustrine species (largemouth bass) and a fast-water specialist (greenbreast darter).

the other two. Central to these issues is the concept of the habitat bottleneck, defined as a limitation of a key habitat type that affects the population dynamics of one or more important life stages of a species such that the limitation is evident at the adult population level. Although relatively simple in concept, there are numerous connotations associated with habitat bottlenecks (Bovee et al. 1994):

1) not all meso- or microhabitats are equally important to the survival of a species,
2) the same habitat bottlenecks do not apply constantly (or consistently) across years,
3) the same habitat bottlenecks do not apply constantly (or consistently) across streams,
4) the same events that cause bottlenecks during one part of the year may not cause bottlenecks at other times,
5) some bottlenecks are associated with short-term (acute) events whereas others are long-term (chronic).

Research conducted during the past decade suggests that habitat bottlenecks are often associated with early life history phases of fish species. Critical microhabitats regularly include those for spawning and incubation, rearing areas for newly emerged fry, and optimal feeding/predator avoidance areas for fingerlings (Nehring and Anderson 1993; Bovee et al. 1994). Typically, the events that cause bottlenecks during early life history are acute. That is, the event is usually one of short duration, lasting anywhere from several days to several weeks.

It is reassuring to find strong relationships between habitat and population dynamics. Even if we know the habitat bottlenecks and critical habitat types affecting a population, however, they may all change as soon as we modify the operation of the system. If we relieve one habitat bottleneck, another will sooner or later take its place.

There are three basic approaches for handling the uncertainties associated with habitat bottlenecks: ignoring them, addressing current bottlenecks, and forecasting future bottlenecks through the use of population models. By ignoring bottlenecks, the strategy used in the assessment of environmental impacts is to treat all microhabitats equally. That is, a balance is struck in the microhabitat changes (positive or negative) that occur for all life stages and species. Most biologists will empathize with the desire for such a conservative approach because they understand the uncertainties inherent in biological systems. Engineers, project developers, or other stakeholders in an IFIM project are not likely to be so understanding. Increased conservatism on the part of habitat management interests usually translates into decreased flexibility with respect to viable alternatives. Consequently, what seems like prudence from a resource manager's perspective may look like intransigence to the other stakeholders.

To overcome the dilemma of the conservative approach, some investigators will build a biological feedback mechanism into their studies. The simplest way to do this is to identify the habitat bottlenecks that currently limit populations of the target species. A more involved process is to use an effective habitat time series, which is essentially a simplified population model that is responsive to changes in habitat availability for a target species. Several investigators have attempted to integrate sophisticated population models with traditional habitat time series analysis (Cheslak and Jacobson 1990; DeAngelis et al. 1990; Williamson et al. 1993). Population models may enable managers to forecast new habitat bottlenecks (or the relief of old ones) that will develop as a consequence of a proposed action. The use of any kind of population model, even one as simple as the effective habitat time series, however, will add considerable complexity to the instream flow study. The important consideration during the identification of issues is whether decisions will be based on changes in habitat or on changes in populations (i.e., what is the currency of the negotiation?). This decision is not trivial. The kind of biological information needed to identify habitat bottlenecks or calibrate a population model cannot be derived solely from IFIM. If stakeholders want population-related information from the methodology, a considerable amount of population-related information must first be put into it.

Population data are not cheap, and their acquisition can add years to the lifetime of an instream flow study. Population models can become so complex that only a few specialists on the interdisciplinary team actually understand how and why the models work. Because of the complexity, alternatives analysis using a population model can become a "black box" exercise for the majority of stakeholders. The constraints of cost, time, and complexity often lead stakeholders back to the conservative approach to river habitat management. In many cases, a conservative approach may not be a bad idea anyway. Biological systems do not operate with the precision of an engineering system, so a cautious attitude is probably warranted regardless of (perhaps in spite of) how much we know.

Determining Habitat Requirements and Temporal Variations

Once we have identified appropriate target species for a study and perhaps even determined which types of microhabitat are of greatest biological importance, we are faced with the next hurdle of an IFIM study: defining what constitutes suitable microhabitat for the target species. Defining the suitability of microhabitat is not easy for a variety of reasons.

1) Many riverine organisms, especially fish, inhabit many different microhabitats during their life cycles. Shifts in microhabitat use occur as a function of size (young fish vs. adults), activity (feeding vs. resting vs. spawning),

season (summer vs. winter), and time of day (dark vs. light), among other things (Orth 1987). We cannot simply define microhabitat requirements for the species in general, we must specify the life stages, sizes, activities, and time periods for which our definitions hold. Often, available knowledge of microhabitat requirements will miss the mark with respect to the detail needed for an analysis.

2) In IFIM, microhabitat requirements are depicted in the form of habitat suitability criteria. Because the microhabitat requirements of many species are somewhat plastic or poorly defined, it follows that habitat suitability criteria obtained from one stream may not be valid in another. This inconsistency in habitat use or definition epitomizes the issue of transferability, the evaluation of the validity of criteria developed elsewhere (a source stream) for use in a stream under study (the destination stream).

3) Microhabitat requirements and habitat suitability criteria are subject to human interpretation, which may be less than objective. Some stakeholders will claim that as long as the streambed is wet, there will be plenty of microhabitat available for everything in the stream. Some may not even acknowledge wetness as a prerequisite for suitable fish habitat. Although these are extreme examples, issues relating to the interpretation of microhabitat use invariably arise.

Historically, a variety of approaches has been used to address the issues of acquisition and transferability of habitat suitability criteria. One of the most popular has been to obtain criteria from another source and ignore the issues of quality or transferability. The appeal of this strategy is that it is cheap (in all respects) and it gives the appearance that the study team is absolved from the responsibility of ensuring the quality of the criteria. At the opposite end of the spectrum is the strategy of developing site-specific criteria for every instream flow study conducted. Although this approach might sound good, some streams are inappropriate for development of habitat suitability criteria. Instead of determining what constitutes good microhabitat for a species, criteria developed in a habitat-poor stream may reveal only what the species can tolerate.

The approach that we recommend is a combination of prioritization, acquisition, and testing. A listing of priority microhabitat types/life stages should be prepared and cross-referenced with a listing of available habitat suitability criteria. Missing criteria may be ignored if they are not essential to the success of the study. Conversely, if high priority criteria are not available, measures must be taken to acquire or develop the criteria during the study. Once the issues of criteria acquisition have been settled, the study plan should also contain provisions to evaluate the transferability of the criteria.

Spatial Composition, Configuration, and Continuity

These issues all relate to the spatial distributions of different kinds of microhabitats in a river. McGarigal and Marks (1995) define compositional metrics as different measures of patch sizes and relative proportions in a landscape (or in our case, a riverscape). Configurational metrics are those that describe the physical distribution and spatial arrangements of patches (McGarigal and Marks 1995). Two examples of configurational measures are interspersion and contagion. Interspersion measures the degree of dispersion or fragmentation of patch types in an area, whereas contagion is a measure of the clumpiness of patch distributions. With varying degrees of difficulty, it is possible to derive some measures of composition (e.g., suitable habitat for a life stage or habitat diversity) from the existing microhabitat models in IFIM. In contrast, configurational metrics are spatially explicit and must be derived from a true two-dimensional habitat model. Currently, there is a great deal of activity in the development and testing of two-dimensional hydraulics and habitat models using the power of high technology, such as geographic information systems (GIS), Global Positioning System (GPS), and advanced surveying instruments (total stations and laser range-finders). Although these technologies have not yet achieved the infrastructure and support system needed for routine IFIM studies, they are expected to mature in the next few years (Bovee 1996; Hardy 1996).

Continuity refers to the extent to which organisms are able to move among different parts of the river. In a longitudinal sense, we need to be concerned that the rearing areas for adult fish are connected to the spawning areas by a traversable, survivable length of stream. Often, the most important consideration with respect to longitudinal continuity is to provide sufficient streamflow during migration to ensure passage past natural barriers and to eliminate any potential thermal or water-quality barriers.

Several attributes of lateral continuity may be important in some applications of IFIM, most often when associated with rapid fluctuations in streamflow. The general terminology associated with this issue is "ramping rate." The basic issue is that when the discharge in a stream changes, the microhabitat conditions at a fixed location change; in effect, areas of suitable microhabitat migrate back and forth across the channel. If the rate of lateral migration exceeds the ability of the organism to keep up with it, the organism will either drift downstream (e.g., aquatic insects) or die (e.g., fish eggs). A different form of lateral continuity issue occurs when the organism is enticed into suitable microhabitats during high flows but is stranded there when flows are reduced. Both of these issues can be evaluated quite well within the context of IFIM.

The foremost anthropogenic cause of habitat fragmentation in rivers has undoubtedly been the construction of

mainstem dams and reservoirs. The primary concern over habitat fragmentation is that it prevents exchange of individuals among populations and periodic recolonization of habitats. This must be a major consideration given the extent to which tributary and mainstem populations are isolated by intervening dams. Osborne and Wiley (1992) provided evidence that fish assemblage structure in warmwater tributary streams is influenced by immigration from main channel populations. Sheldon (1987) predicted that large-scale fragmentation will cause local extinctions as populations are isolated from sources of immigrants.

Key Points of Microhabitat Issues

Microhabitat variables commonly include depth, mean column velocity, cover type or function, and substrate characteristics. Other physical microhabitat variables may be substituted into the analysis, provided that they are related to hydraulic or structural characteristics of the stream.

The selection of target species is a necessary part of an IFIM analysis but may become controversial if the choice can be interpreted as a policy statement. Sometimes, the selection of certain target species can appear to be an intentional attempt to manipulate the results of the study.

Microhabitat requirements must be determined for numerous life stage and temporal stratifications within IFIM. Issues related to habitat suitability criteria include the identification of critical habitats and potential habitat bottlenecks, necessary degree of stratification, and transferability of the criteria to the stream under study.

Microhabitat composition, configuration, and continuity are all issues related to the patchiness and connectivity of habitats at various scales. Some compositional measures, such as areas of suitable microhabitat, are generated easily within IFIM's component models. Other metrics, such as habitat diversity, must presently be calculated externally from any of the IFIM component models. Configurational metrics cannot presently be simulated in the IFIM component models because such metrics must be derived from true two-dimensional models.

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Chapter 3. IFIM Phase II

Study Planning

To people unfamiliar with IFIM, the early stages of a study may seem to lack organization and structure. This perception is more illusory than real, but it originates from the fact that IFIM is designed for maximum flexibility. Investigations can be tailored to fit almost any instream flow problem or analysis of riverine habitat. Study designers are allowed great latitude in the variables they choose to include in or exclude from an investigation. Even the location and amount of data to be collected are left largely to the discretion of the investigators. Innovation and creativity are encouraged, and elegant solutions to complex problems are sought. The ad hoc nature of IFIM planning can appear to be somewhat chaotic to people who are accustomed to more rigid and inflexible processes.

Creativity and innovation should not be mistaken for chaos. Order is maintained in IFIM by adhering to certain procedures during study design. Although there is no precise recipe for conducting a particular kind of IFIM study, there are standard ingredients common to all IFIM studies. As IFIM has evolved, our concepts of how study planning should be conducted have coalesced into the . . .

10 Essential Components of a Plan of Study

1) A comprehensive description of the proposed action and a characterization of the stakeholders and issues.
2) Identification of target species or valued natural resources.
3) Selection and rationale of a methodology to address the issues.
4) A concise statement of study objectives.
5) Study area and segment boundaries.
6) Identified baseline or reference conditions.
7) Details of geographical coverage, data collection, calibration, and quality control for IFIM models.
8) Assignment of responsibilities and authorities.
9) Schedules of activities, milestones, and deadlines.
10) Reconciliation of resource needs with resource availability.

The first and second components can be satisfied by summarizing the results from Phase I. Our goals in this chapter are to discuss the remaining eight components in sufficient detail that the reader can participate intelligently in the development of an IFIM study plan. Some of the components can be logically grouped, and, where practical, we will do so in ensuing discussions. For example, the issues and values of the stakeholders will largely determine what type of a problem you are dealing with. The type of problem dictates the type of methodology that can be employed to solve it. Selected target species or valued natural resources often manifest themselves in the study objectives. Establishing baselines and geographical boundaries frequently share common features. Reconciliation of resources feeds back to the schedule of deliverables, milestones, and deadlines.

Selecting the Appropriate Methodology

The dichotomy between types of instream flow problems and technologies is discussed thoroughly in "The Instream Flow Incremental Methodology: A Primer for IFIM" (Stalnaker et al. 1995) and will be mentioned only briefly here. We have placed political and environmental problems into one of two categories, standard-setting or incremental, depending on the objectives of the decision process. In a standard-setting problem, the analyst is called on to recommend an instream flow requirement, below which water cannot be diverted, to guide low-intensity decisions in preliminary planning and project feasibility studies (Trihey and Stalnaker 1985).

An incremental problem refers to a high-intensity, high-stakes negotiation over a specific development project. The term incremental implies the need to answer the following question: What happens to the variable of interest (e.g., aquatic habitat, recreation value) as a result of a proposed action? In IFIM studies, the types of proposed actions we typically deal with are those that will change the flow regime, the shape of the channel, the thermal regime, the amount of shading along the stream, or the loading of sediment or pollutants to the stream.

At the completion of Phase I, you should have a fairly clear idea of the type of problem you will be dealing with simply by determining whether the study must be able to address alternatives and competing proposals. Standard-setting techniques are inappropriate for brokered decisions, which require the exploration of alternatives. Standards are by definition non-negotiable. Furthermore, standard-setting methods address minimum flow issues only. If the problem revolves around other habitat variables or any other aspect of the flow regime besides low flows (the magnitude and timing of high flows, for example), IFIM might be necessary even though the problem is presented in a standard-setting context.

Attributes of Good Study Objectives

Objectives, as taught in planning courses, are subunits of goals. A goal is what is to be achieved overall, or at least a target to be aimed for. Objectives should be precise, measurable, and achievable. When the objectives have been met, they should indicate progress toward or achievement of the goal. In other words, a good objective will not only tell you what you intend to do, it will also let you know

when you have done it. For example, the goal of a project might be to enhance rainbow trout production in a river reach that is subject to some potential change. One of the study's objectives might be to improve habitat conditions affecting early life history and recruitment of rainbow trout.

Perhaps the best way to learn about setting usable objectives is to share experiences about the successes and failures of objectives in previous studies with professional colleagues. This section aims to augment that invaluable experience by describing the balance between having too much and having too little detail in study objectives.

Bad objectives may defy definition, but we can readily recognize them when we see them. What makes bad objectives so recognizable is that most of us have seen a lot of them during our careers. We may have even written a few. Sadly, we cannot construct a good objective merely by revising the syntax of a poor one. Good objectives usually have a number of recognizable attributes, one or more of which is usually missing from poorly constructed objectives. A good objective is specific, it encapsulates motives, and it defines the currency of the negotiation. A good objective is feasible, specifies deadlines and performance criteria, and incorporates flexibility to allow shifting direction if necessary.

Alice meets the Cheshire Cat

"Cheshire-Puss," Alice began... "Would you tell me, please, which way I ought to go from here?"

"That depends a good deal on where you want to get to," said the Cat.

"I don't much care where---" said Alice.

"Then it doesn't matter which way you go," said the Cat.

"---so long as I get somewhere," Alice added as an explanation.

"Oh, you're sure to do that," said the Cat, "if you only walk long enough."

- Lewis Carroll
Alice's Adventures in Wonderland

Specificity

It is important not to lose sight of the larger study goals because of too much attention to detail. The more common problem, however, is to set objectives which are not specific enough to guide the project where it needs to go. As the Cheshire Cat told Alice, if you do not know where you are going, any road will take you there. A series of case studies of hydropower license negotiations was researched by scientists at the National Ecology Research Center (precursor to the Midcontinent Ecological Science Center, U.S. Geological Survey, Biological Resources Division) during 1992 and 1993. One of the central findings in these case studies was that in order to achieve a successful negotiated agreement, the technical boundaries of the problem must be clear to all parties-at-interest (Fulton 1992). "No matter what a dispute centers on, the need to specify the boundaries and to designate a time horizon for analysis is overriding" (Susskind and Weinstein 1980). Bingham (1986) notes that parties must agree on the scope and the technical facts of issues if they are to be successfully negotiated. The ability of parties to agree on study issues depends on both the "ripeness" of a dispute and on the technical and scientific complexity of the issues (Harter 1982). If parties cannot develop and agree upon an acceptable body of scientific knowledge in an issue such as instream flow, the chances of resolving the dispute by negotiation are slim.

In our case studies, we found that successful negotiations were sometimes blocked because parties failed to agree on appropriate study objectives. In one case, the scale of the study was in dispute: that is, whether fish passage studies concerned just the facility up for relicense or comprehensive, river-wide passage issues. Failure to agree on the scale prevented the applicant and the resource agencies from reaching final agreement. Frequently, the discussions concerning what was to be studied became major contentions. It was clear from these case studies that a successful license negotiation depended upon agreement regarding scope and timing, technical issues, and interpretation of studies to be conducted, that is, upon the formulation of good study objectives.

Encapsulating Motives

Motives mirror the values and goals of the different stakeholders and, when identified, will go a long way in explaining what each group hopes to get out of the study. In an FERC relicensing project, for example, the motive of the applicant is to get the license renewed, as free of encumbering conditions as possible. The U.S. Fish and Wildlife Service might want to restore habitat to predevelopment conditions for an important fish species. An association of lakefront homeowners might simply want to keep the lake level high enough to use their docks. Objectives are much stronger when they incorporate the motives of all the stakeholders. If stakeholders are reluctant to specify their true motives as an objective, it may be necessary to develop a more neutral motive, such as being able to compare several different management alternatives. The opposite of the forthright expression of motives is the hidden agenda, which undermines any potential for trust relations.

Currency

Currency refers to the measures by which each of the stakeholders will gauge success or failure. For a utility involved in an FERC relicensing, the currency may be kilowatt-hours of electricity generated or gross income to the company. A State fish and game agency might be more interested in populations of sport fish or hours of fishing opportunity. Habitat conditions for a sensitive guild of fishes and aquatic invertebrates might be the most important

metric for the U.S. Fish and Wildlife Service. Conducting an IFIM study without defining the currency is like playing baseball with no bases. Agreeing on the currencies to be used in a study is fundamental to determining the scope of the investigation. For example, it takes vastly more data and modeling expertise to predict changes in fish populations than it does to predict changes in habitat availability. If a stakeholder insists the currency be fish populations (or worse, economic values of the fishery), you can expect a long and arduous study, with no guarantee of being able to measure the currency at the end of the study.

Feasibility

Valid study objectives must be technically, scientifically, and institutionally feasible. Technical feasibility of study objectives relates to whether the proposed studies can actually be done using currently available, state-of-the-art technologies for instream flow research. Scientific feasibility refers to the defensibility of the results from a study under the scientific peer review process. In litigation, scientific feasibility extends to the defensibility of methods, assumptions, and analyses used in the study under cross-examination. Institutional feasibility relates to whether studies proposed are "red flags" to other parties with whom you are negotiating. Some red flags are unavoidable (e.g., when a threatened or endangered species might be affected). Other, avoidable red flags include items such as studying the desirability of recreation activities or species which have never occurred in this location before.

Deadlines and Performance Criteria

A cornerstone of negotiation theory is the importance of milestones and deadlines. If there are no recognizable deadlines, the study may take on a life of its own and can become its own worst enemy with respect to resolving the problem. In our case study research, we found instances where the lack of agreed-upon deadlines was used as a delaying tactic by licensees and a form of job security for the investigators. As long as the studies were incomplete there would be no modifications to the operating rules, which was fine with the licensees. And if the license was not issued, the study would continue, providing long-term employment for the investigators.

Participants in IFIM studies are encouraged to be creative and innovative instead of defensive and confrontational. Sometimes the creative atmosphere can be achieved and sometimes it cannot. Owing to the ad hoc nature of IFIM problem solving, however, there is no absolute right or wrong way to conduct a study. This means the bounds of acceptable study performance must be agreed to in advance. Everyone engaged in setting the study objectives must be able to recognize "good enough" when it is achieved.

Flexibility

The objectives of the various parties to an IFIM study will probably be quite different. Indeed, if everyone had the same objective, an application of IFIM might not even be necessary. Consequently, it may be impossible to write a single, all-encompassing objective for a study involving IFIM (unless it is very long and imaginatively punctuated). It is all right to have multiple objectives for an IFIM study; in fact, they are expected. If the study is to address the legitimate interests of the different parties to the negotiation, it will require the buy-in and commitment from all parties involved. By addressing the principle resource needs of each party in the study, everyone will have something at stake, creating a mutual interest in the successful completion of the study.

The parties establishing the study objectives should also have some agreed-upon criteria for reopening the question of the study objectives. Study objectives should be defined broadly enough so that innovative, creative solutions are not precluded. Fisher and Ury (1981) emphasize the power of an elegant solution: a solution that meets the needs of many of the parties to the negotiation. Elegant solutions are usually not very evident at the start of a study, but they may become apparent somewhere in the middle. The reopening clause allows the flexibility to change the study in midcourse to pursue an innovative solution.

Agreements

It is important for all parties to understand what they have agreed to and to *document* their agreements. Agreements can be destroyed if only one party breaches their good faith, the result of which can seriously set back an instream flow study. In one FERC license case study, a comprehensive river corridor, fish passage study agreement was reached among the parties to the consultation, but the agreement was not written down and signed. One of the parties, because of an intervention on a different project license, decided to drop the river corridor agreement and restrict the fish passage study to the dam currently under relicense negotiation. The other parties to the consultation were left with a disappearing agreement. The lesson from this case study is to know when an agreement has been reached (and what it was) and *write it down*!

Bounding the Problem

Study Areas

Geographical boundaries define the scale and variety of alternatives that can be evaluated. The significance of various habitat impacts and mitigation recommendations often depends on the geographic context in which they are evaluated. For example, a 0.1-cms minimum flow in a headwater stream might be extremely important to the local cutthroat trout population but meaningless to habitat restoration goals for an endangered species 500 km downstream.

The first decisions related to geographic boundaries regard the number and aggregate length of streams incorporated

in the habitat analysis. Site-specific impact studies (Fig. 3-1a), often associated with by-pass flows at small hydroelectric facilities, are the simplest and most straightforward. Typically, only a small portion (1 to 10 km) of a single stream makes up the study area. In a linear network (Fig 3-1b), a single stream is divided into two or more pieces, called segments, to reflect longitudinal changes in channel form, hydrology, temperature, or water quality. The aggregate length of stream involved in a linear network may range from tens to hundreds of kilometers, but the key feature is that the analysis is concentrated on a single stream. A parallel network (Fig 3-1c) usually consists of at least three segments: two on tributaries to a stream and one on the mainstem below their confluence. Parallel networks may be smaller than linear networks (commonly less than 100 aggregate river km). The level of complexity of a parallel network, however, is considerably higher because all baselines and alternatives must be developed and integrated for three or more places in a river system, many of which are commonly codependent. The most complex IFIM studies take place in composite networks (Fig 3-1d), which contain linear and parallel elements. Composite networks contain at least three parallel and two linear elements, and may consist of thousands of aggregated stream kilometers.

Study Area Boundaries

The upper end of the study area is usually bounded by the location of the project where the proposed action will take place. The exceptions to this guideline include:

1) upstream reaches that will be made inaccessible to migration, resulting in total loss of habitat above the project,
2) upstream reaches that might figure into the mitigation plan for the project, and
3) projects involving transbasin diversions.

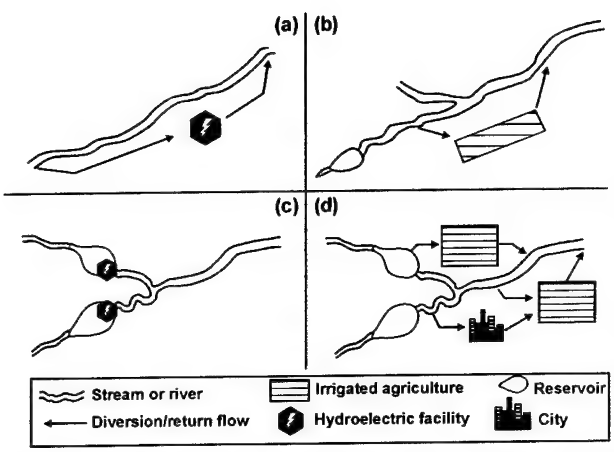

Fig. 3-1. Configurations and typical scales of study areas commonly encountered in applications of the IFIM: (a) site-specific, (b) linear network, (c) parallel network, (d) composite network.

In theory, the downstream study area boundary should be placed where the effect of the proposed action is no longer detectable. However, it is often difficult to detect where one effect ends and another starts, so this approach is impractical in some studies. For example, the impacts of hydropeaking operations on physical microhabitat are more severe closer to the dam. Farther downstream, the pulses from the hydropeaking operation are attenuated; their effects will be noticeably less than they were just downstream from the dam. In contrast, temperature and water quality characteristics may become more important at some distance downstream from a project or a point of withdrawal. The most difficult situations evolve when there is a debate over whether the downstream effects are actually detectable (e.g., cumulative impact assessments that include high watershed projects that may or may not affect habitat hundreds of kilometers downstream).

Owing to impracticalities of applying the "correct" procedure, most practitioners of IFIM follow more relaxed standards when establishing downstream study area boundaries. Often, the lower boundary can be conveniently located where the study stream converges with a large reservoir, another river, or the ocean. The effects of a proposed action might extend beyond the lower boundary, but the study area can be terminated on the assumption that the greater impacts will occur above the boundary. Perhaps the best guidance we can offer is to restrict the study area to the portion of stream where the impact of a proposed action, or opportunities for mitigation, will be greatest. If the study area is too large, impacts occurring in small components may be obscured.

Stream Segments

A strong philosophical tenet of IFIM is the idea that alternatives are evaluated by comparing the total amounts of habitat under baseline and various alternative management conditions. The segment is the basic habitat accounting unit used in these comparisons. A segment is a relatively long section of stream, typified by a geographically homogeneous flow regime. The discharge at the top of the segment at any time of the year should be about the same as at the bottom. The overall channel geomorphology (slope, sinuosity, channel pattern and structure, geology, and land use) is also usually consistent within segment boundaries.

Flow regime is the primary determinant of segment boundaries. Because so many applications of IFIM involve manipulating the water supply, it is essential that the same water supply is applied to each segment. Otherwise, keeping track of what is happening whenever we change the flow regime becomes very difficult. A common rule-of-thumb is to insert a segment boundary wherever the base flow changes by 10% or more (Bovee 1982). The 10% criterion was selected because stream gaging error often makes it

difficult to detect smaller changes in discharge. However, it may also be difficult to detect relatively large differences in base flow where the primary source of accretion is from small tributaries and groundwater inflow. Where there is a great deal of nonpoint accretion of streamflow, segment boundaries can be established where 10% or more additional drainage area accumulates.

Segments may be subdivided on the basis of slope, channel morphology, or valley orientation. These subdivisions are probably more important when macrohabitat features such as water quality or temperature are to be incorporated into the total habitat model. Valley orientation is especially germane in temperature modeling because it influences the amount of shading the stream receives from canyon walls or vegetation during the day. Subdivisions based on slope or channel morphology are important only if there are substantial differences in microhabitat characteristics and macrohabitat suitability between the top and bottom of the segment (e.g., where temperatures are suitable in the upper portion of the segment, but physical microhabitat is not).

Definition and Identification of Baselines

Baselines serve as the benchmarks for developing and evaluating alternatives. They establish the reference points against which comparisons are made. Hydrologic, thermal, water quality, and biological baselines have many common attributes and are grouped together as "time series data." We will discuss some of the characteristics desirable in a time series baseline and will illustrate some techniques for evaluating these characteristics.

The geomorphic baseline is unique. Appropriate time-steps might be measured in decades and periods of record in centuries for some geomorphic processes. Although the same kinds of attributes are desirable for geomorphic and time series baselines, the way the geomorphic baseline is analyzed is vastly different. For this reason, a separate discussion of the geomorphic baseline follows the more generic description of time series baselines.

Time Series Baselines

Time series data, as the name implies, are continuous chronological records of a variable (although there may be gaps in the record). The length of the time series is called the period of record. For a variety of reasons, it is common practice in applications of IFIM to select only a portion of the total period of record for a baseline. Determining which portion of the record to use is a very important, and sometimes controversial, part of study planning. In addition, time series data are often averaged over different time intervals. The averaging interval for the time series is known as the time-step (e.g., daily discharges are commonly measured every 15 min and averaged over 24 h). Because the time-step represents a distillation of information, another important part of study planning is to determine the appropriate time-step to be used in the analysis.

In applications of IFIM, the baseline is usually intended to represent existing conditions of water use and management, prevailing thermal regimes, and water quality. In some cases, however, a baseline that predates existing conditions is used to represent the system under "natural," or at least "predevelopment," conditions. In either case, it is important not to combine predevelopment conditions with postdevelopment conditions in the same baseline. Such a combination is known as a periodicity shift. Likewise, it is desirable not to incorporate trends into the baseline, because to do so may result in miscalculating the water supply or misinterpreting temperature and water quality data under existing (or "natural") conditions. A stationary time series is one that does not incorporate trends or periodicity shifts.

In most rivers, the longer the period of record, the greater the odds that the time series will not be stationary. This phenomenon presents a dilemma to study planners because it is usually desirable to use a relatively long record. The problem involves selecting an appropriately long period of record with no trends or periodicity shifts but with the right time-step to adequately represent either an existing or natural condition.

The period of record. Planners and managers generally agree that when it comes to time series baselines, the longer the better. There are several good reasons to try to represent time series baselines with a long period of record. From the ecological perspective, one hypothesis holds that populations and communities are regulated by extreme conditions that occur within and among years, rather than by ordinary or average events (Wiens 1977; Connell 1978; Grossman et al. 1982). Thus, while water users often plan around the average hydrologic condition, water managers must plan for extremes in the water supply (Dunne and Leopold 1978). At the macrohabitat level, sanitary engineers design waste treatment facilities to maintain adequate water quality during periods of low flow and under extreme weather conditions, rather than under a more normal state (Velz 1970). For all of these reasons, it is desirable to use a long period of record.

To resolve the dilemma of stationarity, it is helpful to consider the planning horizon of the stakeholders. As a rule-of-thumb, we suggest that the period of record should be about twice as long as the planning horizon. For example, managers in an instream flow study might only consider contingency plans for events with a 10-year recurrence interval. This planning horizon suggests that a record of at least 10 years is necessary to estimate either the 1-in-10 wet or 1-in-10 dry condition. If we want to plan for both 1-in-10 year extremes, a 20-year baseline is warranted.

Rules-of-thumb usually come with caveats, and this one is no different. First, it is important to realize that an accurate estimate of a 10-year event is not guaranteed, even with a

20-year period of record. Thus, we revert to our first rule-of-thumb: when it comes to baselines, the longer the better. The second caveat is that any proposed baseline should be representative of the planning horizon. If our planning horizon is 10 years, the baseline should not contain the flood of record or a drought of dust bowl proportions. In this eventuality, stakeholders should consider omitting the most extreme years or picking another, less extreme portion of the period of record.

Time-steps. Here we distinguish briefly between hydrologic and other types of time series. Time-averaged data are commonly used in hydrologic time series, but time series of temperature or water quality variables are commonly recorded as daily extreme values. Where extreme values are chosen, they will normally apply to a daily or weekly time period.

Time-steps for hydrologic data used in IFIM can range from 1-2 h to 1-2 months. The choice of the time-step is determined by the variance of flow over time, both for the baseline condition and under the proposed alternative. The same time-step must be used in comparisons between baseline and alternative conditions. For discharges to be considered homogeneous within a time-step, we suggest that the coefficient of variation (ratio between the standard deviation and the mean) should not exceed 100%. The more variable of the two time series should be used to establish the time-step.

The use of longer time-steps allows the examination of longer periods of record (although this factor is becoming less important as personal computers become faster and more powerful). Long time-steps may also average out biologically significant variations in streamflow, however, which would defeat the purpose of a long record. It may be helpful to examine the record for correlations between events averaged over short and long time-steps. If these events are strongly correlated (e.g., $r^2 > 0.90$), longer time-steps can probably be used without losing biological relevance.

The most extreme time-step used in applications of IFIM is associated with hydropeaking operations. Releases from these facilities can range from near zero to hundreds of cubic meters per second, literally in minutes. It is common to use a 1-h time-step for hydropeaking applications, although in some situations only the daily extremes are used. The use of hourly time-steps presents a unique problem to the analyst because it is difficult to investigate long periods of record on an hourly basis (just imagine the computing requirements for 30 years of hourly data). In this case, a workable solution to the conflicting needs of hourly and across-years levels of analysis is to examine a subsample from the baseline period of record. Flow duration curves of monthly, seasonal, or annual streamflow statistics are used to identify wet, dry, and normal periods. Then, 1- to 2-week periods are sampled from the stratified database and analyzed on an hourly time-step.

Evaluating time stationarity. A stationary time series is one that does not exhibit trends or periodicity shifts. A trend is defined as a unidirectional change in the average monthly or annual discharge in a time series (Fig. 3-2). A periodicity shift is a change in the variance of streamflows. A trend has occurred, for example, if the total water yield at a gage steadily decreases over a 20-year period. A periodicity shift is indicated if average daily flows became more variable over time, even though the monthly averages remained constant. A special type of nonstationarity is a step-change, whereby the average flow or its variability changes abruptly. Common examples of developments associated with step-changes include new reservoir construction or the installation of a new diversion structure.

All three forms of nonstationarity can be identified by graphical or statistical methods. One of the easiest ways to examine a time series for extreme trends and large step-changes is simply by plotting discharge over time (Fig. 3-2). If a step-change has occurred, the hydrograph will often appear to suddenly shift one way or another. Extreme trends may also be quite obvious; for more subtle trends, though, it is sometimes informative to perform a linear regression with time as the independent variable and discharge as the dependent variable (e.g., Fig. 3-2).

The double-mass curve (Hindall 1991), a cumulative plot of one hydrologic variable versus another over time (Fig. 3-3), is another easy graphical technique for examining trends in a time series. Although the double mass curve in Fig. 3-3 shows a plot of precipitation versus streamflow, one could also plot a hydrologic time series known to be stationary versus the time series for another river. If the time series for the second river is stationary, the double mass plot will appear as a straight line. In the case of the trend illustrated in Fig. 3-3, the slope should be relatively constant over all of the years if the flow trend was related to precipitation.

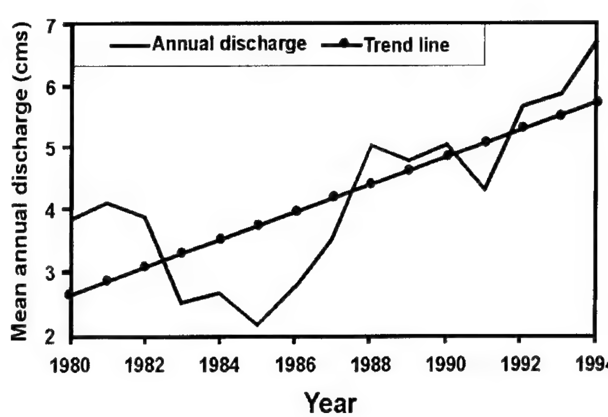

Fig. 3-2. Example of a trend in mean annual discharge.

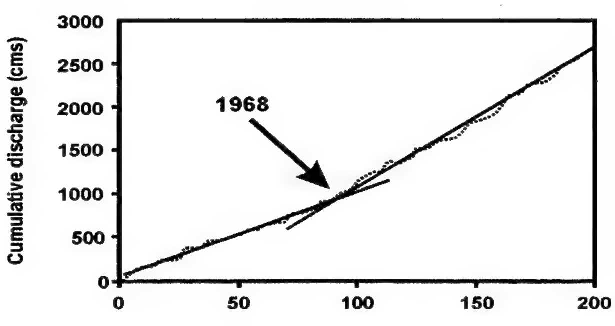

Fig. 3-3. Double-mass curve of mean monthly precipitation and streamflow for the Huron River at Ann Arbor, Michigan.

The break in slope in Fig. 3-3 indicates a change in water yield from the watershed, because the amount of runoff from each precipitation event increased during that time period. (The change in runoff resulted from the 1968 failure of two small flood control reservoirs in the headwaters.)

Statistical techniques can also be used to detect or confirm suspected trends. One of the easiest of the statistical approaches applicable to trend analysis is the Hotelling-Pabst test (Table 3-1), a two-tailed measure of rank correlation.

Table 3-1. Computational table used with the Hotelling-Pabst test for trends. Mean annual discharges are for the Huron River at Ann Arbor, Mich.

(a)	Mean annual discharge (cms) (b)	Year rank $R(x_i)$ (c)	Discharge rank $R(y_i)$ (d)	$[R(x_i)-R(y_i)]^2$ (e)
1960	14.9	1	11	100
1961	9.7	2	6	16
1962	9.8	3	7	16
1963	7.4	4	2	4
1964	5.3	5	1	16
1965	8.7	6	4	4
1966	8.1	7	3	16
1967	10.6	8	8	0
1968	19.4	9	14	25
1969	19.4	10	15	25
1970	12.0	11	10	1
1971	11.9	12	9	9
1972	9.5	13	5	64
1973	19.3	14	13	1
1974	23.5	15	17	4
1975	16.1	16	12	16
1976	20.7	17	16	1
			T =	318

The test is conducted by ranking the time-steps ($R(x_i)$) chronologically in a series (column c) and then ranking the flows associated with each time-step ($R(y_i)$) according to magnitude (column d). The test statistic (column e) is given by:

$$T = \sum [R(x_i) - R(y_i)]^2 \quad (4)$$

The null hypothesis of no trend is rejected if T is larger or smaller than its critical quantile values (Appendix). A positive trend is indicated if T is less than the lower quantile value (i.e., T = 0 if $R(x_i)$ and $R(y_i)$ are ranked identically). If T is greater than the upper quantile value, the trend is negative. In Table 3-1, the critical quantile values are 420 and 1,217 for n = 17. The calculated value of T was 318, which is less than the lower quantile value. Because the value of T is smaller than the minimum quantile value, we can conclude that there is a trend in the data (at the 0.05 level of significance) and that it is positive.

A periodicity shift may appear as a trend in flow variability when factors affecting watershed mass-balance processes are involved (e.g., reforestation following fire or timber harvest). The same tools used to identify trends can be used to analyze a record for a periodicity shift. The only difference between the two types of analysis is that a mean value of the variable is used to examine trends, whereas periodicity shifts are evaluated using the variance.

When analyzing stationarity of a hydrologic series, it is important not to confuse persistence with a trend. Persistence is defined as the nonrandom association of successive members of a time series. In plain English, this means that wet periods tend to follow wet periods and dry periods follow dry periods. Persistence occurs to some extent at all time intervals but may be most notable in annual data. Owing to the phenomenon of persistence, hydrographs of annual discharge often appear to be cyclic. (Some speculate that these cycles are related to sunspot activity or world-wide meteorological cycles, such as El Niño.) Persistence in hydrologic records can be problematic when only part of a cycle is included in the baseline. The nature of the problem depends on which portion of the cycle is included.

There are likely to be trends or periodicity shifts in nearly any hydrologic record in the United States. Whether you need to worry about it depends mostly on the slope of the trend line but also somewhat on the length of your baseline period of record. When comparing the first part of a hydrologic time series with the last part, as a rule-of-thumb, the difference in the overall water supply should not exceed 10-15%. For a 30-year period of record, this means that the slope of the annual trend line (e.g., Fig. 3-2) should represent no more than about a 0.3% accretion or depletion per year. Naturally, the 0.3% per annum criterion is just a guideline. How much of a trend is tolerable in a study is one of

those performance criteria we mentioned earlier. It is up to the study group to decide how much is too much.

What can be done to correct the problem if the group decides that the baseline time series is not sufficiently stationary? The easiest solution is to lengthen, shorten, or pick another portion of the period of record for the baseline. A trend-free time series may be less important than one that is representative of existing and future water supplies and uses. If an unacceptable trend exists or it is impractical to adjust the period of record, however, the time series can be detrended. Detrending a time series involves correcting for changes in water use, watershed mass balance, or precipitation regime and can be performed by the group's resident hydrologists. Because there may be numerous ways to detrend the series, however, we recommend that hydrologists from several stakeholder groups work together as a hydrologic task force to derive a single, mutually agreeable time series.

Hydrologic Cycles versus Persistence

The 83-year time series for the Huron River at Ann Arbor, Michigan, exhibits several distinct hydrologic cycles (Fig. 3-4). This time series contains three wet and two very dry periods. The portion BCD of the hydrograph in Fig. 3-4 represents a wet cycle. If this were the only portion included as the baseline hydrograph, the water supply would have been seriously overestimated. Similarly, if only the portion DEF were used, the water supply would have been underestimated. Using portions ABC, CDE; or EFG, the analyst might conclude that there is a trend in the series. In reality, these portions represent full amplitude half-cycles and might be very good representations for a hydrologic baseline used in IFIM.

Where cyclic phenomena are evident, it is advisable to balance the number of wet and dry cycles. Portions CDE and EFG each contain one wet and one dry period, but if both portions were used (CEG), the hydrologic baseline would contain two wet and one dry period. The portion CEG could lead to the same problem of overestimating water supplies as using portion BCD. It would be much better to use A-G or C-H because they contain two wet and dry cycles.

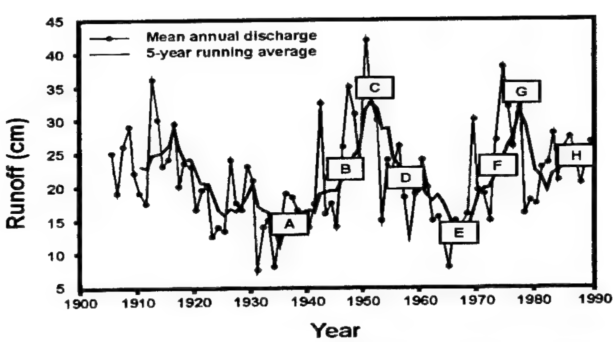

Fig. 3-4. Mean annual discharges and 5-year running means for the Huron River at Ann Arbor, Michigan, 1907-90. The letters A-H represent various half cycles within the hydrologic record.

Geomorphic Baselines

Geomorphic baseline refers to watershed characteristics, land use, and water management practices that affect the structure, pattern, and stability of the river channel and floodplain. The importance of stationarity for the geomorphic baseline in habitat assessments is often not as widely recognized as for the hydrologic, thermal, and water quality baselines. This is unfortunate because many riverine habitat problems are more directly associated with channel changes than with water management practices.

In the vast majority of IFIM applications, the geomorphic baseline is established by measuring cross-sections of the river as it exists today. This practice conforms to our initial definition of a baseline as a measure of the existing condition. There are two scenarios, however, under which the existing channel morphology is insufficient or misleading as a baseline. The first case is represented by an existing channel that is not in a state of dynamic equilibrium. If the channel is undergoing a fundamental change in structure and pattern, today's channel configuration may not be very representative of its morphology in the future. The second case is represented by an equilibrated channel that is inherently unstable and changes shape throughout the year. In this section, we discuss symptoms of channel disequilibrium and instability, as well as options for handling channel change during the study planning phase.

Symptoms of channel change. In instream flow studies, we are primarily concerned with five types of channel change: aggradation, degradation, channel enlargement, channel reduction, and seasonal scour and fill cycles. For planning purposes, recognizing that a change is ongoing is probably more important than diagnosing the exact type of change or its underlying causes. Scour and fill cycles must be distinguished from channel disequilibrium, however, because the planning options will be different.

It may be possible to detect recent channel changes from historical data and monitoring. Old aerial photos are especially useful in detecting changes in channel pattern that accompany structural changes. A very useful source of aerial photography data is the National Aerial Photography Program (NAPP). NAPP is an interagency Federal program coordinated by the USGS. The program was established in 1987 to coordinate the acquisition of aerial photographs for the United States. Taken from aircraft flying nominally at 6,096 m (20,000 ft) above the terrain, each NAPP photograph covers about 83 km^2 of ground. Black-and-white and color-infrared photographs are available from NAPP. The goal of NAPP is to photograph each flight line every 5 years. Although complete coverage along a flight line may not exist due to poor visibility or flying conditions, it may be possible to find enough corresponding ground points to assess channel changes on a 5-year cycle. For additional information on NAPP, please consult

the USGS/NAPP home page at http://edcwww.cr.usgs.gov/napp/napp_examples.html.

Information related to recent channel modification can also be obtained from the Water Resources Division of the USGS. The discharge records for permanent USGS gaging stations are derived from rating curves for the station. If a channel change occurs, the rating curve will no longer be valid and the gage must be recalibrated. In a degrading stream, the local base level of the stream is lowered, resulting in a corresponding reduction in water surface elevations. The cross-sectional area increases in an enlarging channel, allowing conveyance of a greater volume of discharge for a particular river stage. In either case, the stage for a particular discharge will be lower on the recalibrated rating curve than it was on the previous one (Fig. 3-5). Conversely, recalibrated rating curves exhibiting a trend of upward corrections are typical of streams undergoing aggradation or channel reduction. The average streambed elevation usually increases during aggradation, and although the channel commonly becomes wider, the stage for a given discharge will be higher on the recalibrated rating curve than it was on the previous curve. The cross-sectional area of a shrinking channel will be decreased, necessitating a similar upward adjustment of the stage-discharge relation.

Anomalies in rating curves can also be used to identify scour and fill cycles in streams. Seasonal scour and fill cycles occur most frequently in sand-bed rivers, but they may also occur in streams armored by a thin veneer of gravel or cobble. In these streams, the streambed becomes fluid at discharges at or near bankfull, and the cross-sectional area expands dramatically. The stage at a high discharge under conditions of a fully mobile bed can be equal to or lower than the stage at a lower discharge when the bed is not moving (Fig. 3-6). A similar effect occurs in armored channels when discharge is sufficient to remove the armor layer. Once the protective veneer is removed, the stream will quickly erode the finer underlying materials. Many of the symptoms of scour and fill cycles are similar to those of channel enlargements and reductions. The primary difference is that scour and fill is a cyclic process occurring seasonally.

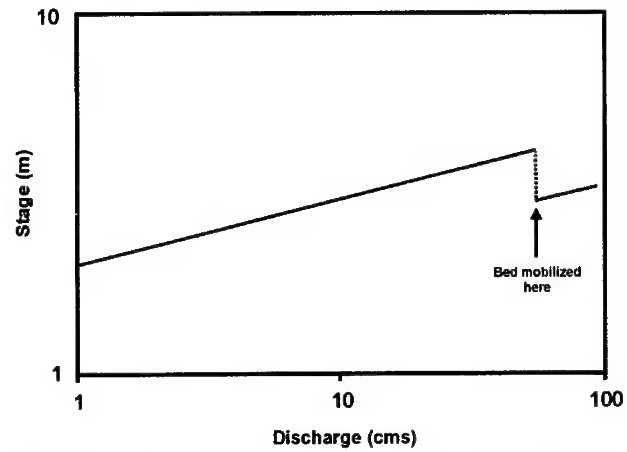

Fig. 3-6. Anomalies in a rating curve associated with scour and fill cycles. Bed mobility is indicated by the "step" in the rating curve between 55 and 60 cms.

Riparian vegetation can also sometimes be used to detect recent episodes of channel disequilibrium. Aggradation and channel enlargement are often accompanied by channel widening. If the channel is expanding laterally, trees on both sides of the river will be undercut and eventually fall into the river. Although trees commonly topple into rivers, two characteristics are indicative of channel widening. First, fallen trees will occur on both sides of the river in straight reaches and not just on the outside of meander bends (Fig. 3-7). Trees that fall into the river only at meander bends are probably symptomatic of meander migration, not channel widening. Second, trees of all ages will topple over under conditions of channel widening. In equilibrium channels, only the oldest and tallest trees would be expected to fall over.

Channel degradation and reduction are accompanied by the abandonment of the previous channel's floodplain and the development of a new one. This process results in the formation of terraces along the river. Vegetation on the terrace will have been established under the flood regime in the previous channel and will be considerably older than the vegetation on the new floodplain (Fig. 3-8). In contrast to channel widening, it is possible to date the establishment of trees on the new floodplain through dendrochronology. This may allow the investigator to pinpoint how recently the trees were established.

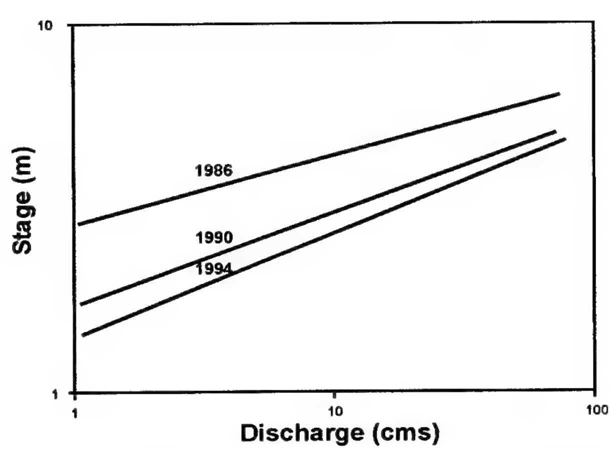

Fig. 3-5. Systematic recalibration of USGS rating curves, typical of streams undergoing degradation or channel enlargement.

Options for accommodating channel changes. Channel disequilibria pose the same kinds of problems that arise

Fig. 3-7. Bank undercutting and trees falling into channel in the upper Tallapoosa River in northeastern Alabama. Aggradation is suspected because trees of all ages are toppling into the channel and a large sand deposit has formed in the center of the channel.

Fig. 3-8. Terrace formation along the Missouri River below Fort Peck reservoir in northeastern Montana. The terrace, with its older cottonwoods, is visible in the background. Young cottonwoods have colonized a new floodplain at a lower elevation in the foreground.

from trends in hydrologic time series. Unfortunately, it is not as easy to detrend a disequilibrated channel as it is to select another period of record or correct for a hydrological trend. When faced with an ongoing disequilibrium process, study planners have four options:

1) Ignore it. In some cases, it may be completely acceptable (with the study planning group's approval) to ignore channel disequilibrium, even though it is known to exist. This is particularly true if the stream is approaching a new equilibrium condition but just has not arrived there yet. Notice that in Fig. 3-5, the largest adjustments were made relatively early in the disequilibrium cycle. If you have access to the type of adjusted rating curves illustrated in the figures, you may be able to tell how close you are to a new equilibrium. If the adjustments made by the USGS are smaller or separated by increasing time periods, the stream is probably approaching a new equilibrium and it is safe to proceed with the study.
2) Wait. The same type of evaluation could indicate that the best thing to do is nothing. That is, if the stream is nearly re-equilibrated and the results of the study are not needed immediately, maybe the best approach is simply to wait a year or two to initiate the study. Doing nothing is a more viable option when the disequilibrium episode is short-lived.
3) Plan a do-over. Where the disequilibrium is gradual and long-term, the channel may not achieve a new equilibrium condition for several decades. From year to year, however, the channel may not change very much. In this situation, it might be possible to proceed with the study, using the existing channel as the baseline condition. A channel monitoring program should be incorporated into the study plan, with the understanding that recommendations from the present study will be revisited every 10 years or so. If the channel changes sufficiently during the time period that the recommendations would change, then a new study should be commissioned.
4) Predict what the new equilibrium channel will look like. Some channel changes cannot be ignored, waited out, or revisited. In particular, planners must be concerned when the source of the channel change is the proposed action itself. In the vast majority of endangered species problems we have investigated over the past two decades, channel change has been a significant problem. When faced with this type of problem, the only way that baseline and postproject habitat can be compared is by predicting what the new equilibrium channel will look like. As mentioned in the introduction, our ability to make this kind of prediction is relatively crude. This crudity does not mean that it is impossible, however, and we will discuss the types of modeling efforts required in the next chapter.

The Scope of Work

During the problem identification phase, habitat variables were identified that were likely to experience biologically significant changes as a result of some proposed action. The issues that were recognized during problem identification are translated into objectives of the study plan. The objectives, in turn, direct our attention to the information needed to be able to formulate and evaluate alternatives. Project scoping addresses the issues related to accumulating the requisite base of information.

In IFIM, the generation of information relies heavily on a combination of empirical data and model output. The primary advantage of mathematical modeling is that models can be used to quantify the effects of proposals that have not yet been put in place or of conditions that cannot be measured. Models are particularly useful in identifying second-order, or chronic effects, where the impacts to target organisms are subtle or extend over longer time horizons.

Second-order effects are illustrated by the linkages between streamflow and dissolved oxygen. Reduced flow results in less water volume, lower surface area, and reduced velocity. Owing to the reduction in volume, the existing load of biological oxygen demand (BOD) is concentrated. The reduced volume, surface area, and velocity result in elevated water temperature. At higher water temperatures, the reaction kinetics of organic decomposition are accelerated, but the solubility of oxygen in water is reduced. As the solubility and velocity are reduced, the rate of mechanical reaeration is reduced. Cumulatively, all of the variables that are affected by the reduction in discharge can change just enough to create a situation where dissolved oxygen is depleted below lethal levels.

Physical process models are mechanically based, an attribute that allows the investigator to determine the model's sensitivity to different input variables. Data collection can thus be prioritized so that the most important data is collected first and most accurately. Modeling is an extremely economical way to amass large quantities of information. Modeling also carries a premium, however, requiring users to understand the theory and fundamentals of the models used as well as the data required to run them. There may be nothing more dangerous in the decisionmaking arena than relying on "black box" model applications.

Whereas problem identification is predominantly an exercise in determining what information is needed to resolve a problem, determining the scope of the study focuses largely on identifying which models will be needed, what data the models require, and where and how much of it will be collected. Inevitably, the scoping process requires a rudimentary understanding of how the IFIM component models operate and of their data requirements.

The acquisition of information and data for IFIM component models typically consists of three related activities:

(1) determining what data are needed and where, (2) conducting an inventory of data already available, and (3) devising a means of filling in the information gaps. This chapter briefly describes these data requirements and sources of information. Filling in the gaps largely defines the scope of study implementation, to be described in the next chapter.

Hydrology

Data Requirements

Hydrologic data requirements should be fairly self-evident by the time you have identified the issues, segmented the study area, and defined the baseline period of record. To complete an analysis, a measurement or estimation of the discharge is needed in every segment for every time-step during the baseline period of record. In order to compare alternatives, it is also necessary to determine what the baseline would be like with the proposed action in effect. Sometimes, the only information needed to generate an alternative flow regime below a proposed diversion is when and how much water will be diverted and any special contingency rules affecting the withdrawal. In other cases, acquiring an alternative flow regime will be a major undertaking, especially if it involves developing or modifying reservoir operations rules or analyzing detailed water rights issues.

Sources of Data

The most widely available source of baseline hydrologic data is the Water Resources Division of the USGS. USGS data are available from most Federal depository libraries. A list of Federal depository libraries can be found on the Internet at http://www.access.gpo.gov/su_docs/libpro.html. This site is searchable if one has the name of the document needed. Constructing a long period of record from published USGS documents, however, can be quite a challenge. Through September 30, 1960, the records of discharge in streams and contents of lakes and reservoirs were published in an annual series of USGS Water Supply Papers entitled *Surface Water Supply of the United States*. Prior to 1951, there were 14 volumes in the series; one each for parts of the conterminous United States corresponding to major drainage areas. From 1951 to 1960, there were 20 volumes in the series, including one each for Alaska and Hawaii. Between 1960 and 1970, the Survey produced two 5-year series consisting of 37 volumes each. Starting in 1970, hydrologic records were published annually, by state, in a series entitled *Water Supply Records for the State of...*

Citing our own experiences, extracting a long-term hydrologic record from hard-copy requires patience, tenacity, and a willingness to hang around photocopying machines for extended periods of time. A alternative way to acquire hydrologic data is to use a commercially available CD-ROM database (e.g., EarthInfo, Inc. or Hydrodata). The daily flow records for all USGS gaging stations in the United States are encapsulated on CD-ROM and with supporting software can be searched very easily. Full or partial records can be copied to a diskette in several popular formats. The best place to look for these databases is in the libraries of large universities (especially land-grant schools), where the data can be accessed for free or for a small fee. The databases can also be purchased or leased through a subscription from private suppliers. If you have access to the Internet, an inexpensive and convenient alternative is the USGS home page on the World Wide Web at http://www.usgs.gov. Our experience with the Internet connection is that it is very easy to extract flow records for any gaging station in the database.

If you need a hydrologic record with time-steps smaller than 1 day (e.g., for hydropeaking projects), the project operator may be able to provide information on release schedules. With the exception of spills, project releases are usually good approximations of the actual streamflow in the reach of stream within several kilometers below the project. Because of a phenomenon known as pulse attenuation, however, project releases will be a poor estimate of the actual discharge at some distance downstream from the project. The range of discharges experienced immediately below the project will be considerably more extreme than farther downstream. If such downstream areas are included in your study area, it may be possible to obtain the uncompiled (unaveraged) discharge records from the USGS. These records are derived from the stage readings at the gaging station recorded at 15-, 30-, or 60-min intervals. If USGS records are not available for the segment of interest, it may be necessary to collect these data empirically or to use a flow routing model (i.e., fill in the gaps).

Sources of hydrologic information dwindle rapidly if they are not available from the USGS. Sometimes, you may be able to obtain streamflow records from a state's water resources agency, which is usually affiliated with the department of natural resources. (This is also where to find information on water rights.) The U.S. Army Corps of Engineers, the Bureau of Reclamation, and the U.S. Forest Service may also be able to provide hydrologic information on particular streams. If streamflow records are not available for each segment in your study area, or if the periods of record for all the stations are not the same, it will probably be necessary to fill in missing records through a process known as hydrograph synthesis. Methods of synthesizing hydrologic records are discussed in the next chapter.

Channel Geomorphology

There appears to be little middle ground when it comes to handling channel geomorphology in IFIM. The task can either be easy, with fairly accurate results, or difficult, with approximations for results. Four physical scenarios are possible with regard to channel geomorphology during an application of IFIM:

1) the stream is currently in a state of dynamic equilibrium and will remain so with the project in place;
2) the stream is currently in a state of disequilibrium and will not be affected by the project;
3) the stream is currently in a state of dynamic equilibrium but will change in response to the project; and
4) the stream is currently in a state of disequilibrium that will either be exacerbated or reversed as a result of the project.

When determining the data requirements for the channel morphology component, it is important to consider which physical scenario fits the stream under study. Additionally, if a channel change is anticipated, it is important to recall the options for dealing with it that were discussed earlier in this chapter: ignore it, wait it out, proceed anyway, or incorporate the change in the habitat model.

Data Requirements

The data requirement for the channel morphology component is a representation of the channel under its baseline and postproject configurations. This requirement is most easily accommodated under the first scenario, where the stream is in equilibrium and will stay that way with the project in place. In this instance, the investigator can measure channel characteristics directly at selected locations and use one set of measurements to represent both the existing and postproject conditions. About the most complicated variation of this scenario occurs in streams that undergo seasonal scour and fill cycles. For these streams, the channel should be measured for both scoured and filled configurations. Habitat simulations are subsequently stratified according to the appropriate seasons or discharge ranges associated with the respective channel configurations. Other options for handling channels in various states of disequilibrium were discussed under the heading of baselines.

The most difficult situation arises when the project will cause or exacerbate an episode of channel disequilibrium (option 4 above). In this case, the impact of the project cannot be assessed without estimating a postproject channel configuration. Simply monitoring the change after the project is in place does not qualify as planning and can be a disastrous strategy. To assess habitat-related impacts of channel change, it will be necessary to predict what the new equilibrium structure will be like, a procedure that is definitely classified as "filling gaps."

Sources of Data

Data on postproject channel structure may or may not be available, depending on whether anticipated channel changes are intentional or incidental. Channelization and snagging projects are examples of intentional channel changes. Because channelization projects usually result in monotonously uniform channels, it is a fairly simple matter to depict the postproject channel from the engineering specifications for the project. The U.S. Army Corps of Engineers, which has jurisdiction over dredge and fill permits, is the logical source for this type of information.

Channel changes that result incidentally from a proposed action are much more difficult to deal with. It may take a serious investigation just to learn the dimensions of the new equilibrium channel. In addition, the pattern and structure of the new equilibrium channel may be different from that of the existing channel. Where changes such as these are anticipated, the study planning effort moves quickly into the study implementation mode.

Water Temperature

Data Requirements

Two basic approaches can be used to predict changes in water temperatures: regression models and heat flux/transport models. Regression models often take the form:

$$T_w = \alpha + \beta T_{air} + \gamma \ln(Q) \quad (5)$$

where T_w is water temperature, T_{air} is air temperature, α, β, and γ are regression coefficients, and Q is the discharge (Bovee et al. 1994). An alternative formulation includes time in the regression equation:

$$T_w = \alpha + \beta \ln(Q) + \gamma \sin(t) + \delta \cos(t) \quad (6)$$

where t equals Julian date and all other parameters are as above.

The data requirements for a temperature regression model are quite modest. A relatively small amount of continuously recorded air and water temperature data, in conjunction with streamflow data, would be sufficient to construct a temperature regression model. However, temperature regression models are also the most constrained in their application. Regression models, regardless of complexity, are likely to be inappropriate for changed conditions such as altering the shading, release temperatures, or stream width. If wider applications or greater accuracy are required, it is far better to use a physically based water temperature model, such as the Stream Network Temperature Model (SNTEMP; Theurer et al. 1984).

Data requirements for SNTEMP include variables related to heat flux and transport equations, which can be classified into four components: stream geometry, meteorology, hydrology, and water temperature.

Channel geometry data. Fundamental stream geometry measurements for water temperature simulations include elevations, stream distances, stream width, channel roughness, and stream shading. Elevations are important in temperature modeling for (1) calculating the slope resulting in heat from friction; (2) calculating the atmospheric pressure, an important element in heat convection; (3) calculating the depth of the atmosphere through which solar radiation passes; and (4) translating known air temperatures and relative humidities from one elevation to another.

Stream distances are important in calculation of heat transport. Distances translate to travel time and thus exposure time to all of the heat flux conditions. Stream width can be a very sensitive parameter in modeling water temperatures. All of the heat flux activities take place at either the air-water interface or the water-ground interface, both of which are as wide as the wetted stream width. Manning's n is a measure of channel roughness and is used in SNTEMP's heat transport model to estimate average velocities, which are in turn used to estimate travel times.Additionally, SNTEMP uses Manning's n to calculate heat generated through the friction of water against the streambed.

Water temperature can be very sensitive to stream shading, especially for low-flow, high-width streams in midsummer. Shade, as considered here, comes in two forms, riparian vegetative shade and topographic shade from valley walls, cliffs, and stream banks. Both forms prevent daily solar radiation from reaching the water's surface. Shading affects stream temperatures in three primary ways. First, it screens the water's surface from the direct rays of the sun. Solar radiation may account for over 95% of the midday heat input during midsummer (Brown 1970). Thus, it is a dominant factor affecting maximum daily water temperature, often more so than air temperature. Second, shade reduces the amount of the water's back radiation at night, tending to moderate the minimum stream temperatures. Third, shade produces its own long wave (thermal) radiation, which also raises minimum temperatures at night.

Meteorological data. Required meteorological input to SNTEMP include data for air temperature, relative humidity, solar radiation, percent possible sun, and wind speed. Air temperature is the single most important (sensitive) parameter in the absence of other thermal inputs because it is related to many of the heat flux components, especially atmospheric radiation, evaporation, and convection. Relative humidity and wind speed are also important components of the evaporative and convective heat flux. Solar radiation represents the energetic income from which heat is derived. Percent possible sun is used in the SNTEMP model as a surrogate for cloud cover, which in turn, behaves much like stream shading.

Hydrologic data. Hydrologic input to SNTEMP has surface and groundwater components. Total discharge is a measure of the total volume of water to be heated, thereby affecting the heat flux component. Total discharge directly affects top width and travel time and may indirectly affect stream shading as it influences the effective offset of vegetation from the stream. Because groundwater is typically cooler than surface water during summer and warmer during winter, the inflow of groundwater can significantly mediate instream temperatures, particularly if groundwater makes up a large portion of the total discharge.

Water temperature data. Obviously, if you had access to a continuous thermograph record that coincided perfectly with the hydrologic baseline, none of the aforementioned data would be needed to develop the thermal baseline. If only a partial record were available, the thermal baseline could be developed from one of the regression equations presented earlier. You still need to develop a time series of temperatures for the with-project condition, however, which will almost always require the use of a model like SNTEMP. The most common use of water temperature data, therefore, is for model calibration and verification. Without this type of data, the output from a water temperature model is basically an educated guess.

Not all water temperature data are equally useful for calibrating or verifying model output. The most versatile data are obtained from continuously recording thermographs and are collected over a period long enough to allow model outputs to be compared with measurements under a wide variety of meteorological conditions and streamflows. Temperature data obtained from maximum/minimum thermometers can also be used for comparative purposes, especially if monitoring is conducted on a regular basis. Grab sample data are the least useful but are better than no data at all.

Sources of Data

Channel geometry data. Some information for the channel geometry component, such as elevations, distances, latitude, and stream orientation can be obtained from topographic maps. Information relating stream width to discharge, determinations of Manning's n, data for calculating stream shading, and continuous records of water temperature, however, will usually not be available; these data must be collected on-site as part of study implementation.

Meteorological data. As with all parameters needed to calibrate a water temperature model, meteorological data may come from a variety of sources. Air temperature may of course be measured in a manner much like that for water temperature. There are also commercially available CD-ROM databases that contain air temperature data originally from the National Climatic Data Center. Unfortunately, these databases do not contain other meteorological data needed by most temperature models. By convention, the primary weather stations record daily maximum and minimum air temperatures midnight-to-midnight. The mean daily temperature obtained by averaging maximum/minimum values is usually less than 1° C different from the true daily average (Linsley et al. 1975).

Additional sources of meteorological data may include:

Airports
Military installations
National Climatic Data Center
Private weather or data services
U.S. Department of Agriculture
  Experiment Stations
  Farm forecast services
U.S. Coast Guard
U.S. Environmental Protection Agency

U.S. Forest Service Offices
U.S. Forest Service Fire Data Center
U.S. Weather Service
Universities
Utility districts/companies

Hydrologic data. Hydrologic data required for a water temperature model are basically the same as those needed for other IFIM components, with one major exception. In addition to the total discharge in a segment, it is also necessary to compute a groundwater component. Because its temperature is relatively constant year-round, the inflow of large amounts of groundwater can have a profound buffering effect on stream temperatures. However, determining the rate of gain or loss of groundwater from gaging station records may not be very accurate. Therefore, it may be necessary to approximate the groundwater flux empirically. During the study planning phase, we recommend conducting a crude sensitivity analysis to determine whether information on groundwater, beyond what can be extracted from surface water records, is warranted.

Water temperature data. The most valuable type of water temperature data is obtained from continuously recording thermographs. The USGS maintains thermographs at some of its gaging stations; locations can be found on the USGS Daily Values database or from local offices of the Water Resources Division. Local offices of the State fish and game agency are also good sources of water temperature data.

Water Quality

Before we discuss water quality models, it is necessary to introduce some fundamental concepts used in them. Several physical, chemical, and biological components are often grouped under the general heading of water quality (e.g., nutrients, salinity, sediment, turbidity, bacteria, aesthetics, pH, odor, dissolved solids, dissolved oxygen, toxicants, insecticides, herbicides, and metals). These components may be either conservative constituents, meaning that they do not decay through time, or nonconservative, meaning they do decay. The dynamics of both are treated separately in water quality models, but both use mass balance equations including growth/decay and sources/sinks.

The tough part of water quality modeling, and the part that really is still in some degree of flux, is how to handle organics and metals. Organics need special treatment because of adsorption, volatilization, biodegradation, photolysis, and hydrolysis. Adsorption comes into play because organics are weakly hydrophobic and attach to sediments, often resulting in resuspension if the sediments have been disturbed. Volatilization refers to the change of organics from liquid to a gaseous phase. This process is complicated because the rate is dependent on the initial concentrations as well as environmental factors regulating vapor pressure, such as temperature. Biodegradation is the microbial reduction of organics to products of lesser toxicity. Though it sounds good, biodegradation may cause an oxygen deficit. Photolysis is the decomposition of organic contaminants by absorbed sunlight; however, sunlight absorption is tempered by turbidity and plant growth. Hydrolysis refers to the pH-dependent decay of compounds through direct interaction with water. All of these processes are difficult to model.

The important metals include arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, and aluminum. Their dynamics are complicated because speciation from one compound to another is controlled by temperature, ionic strength, and pH. Speciation rates are regulated by solubility (precipitation), adsorption, reduction-oxidation, and dilution.

During the planning phase of an IFIM study (preferably during problem identification) you should look closely at whether the problem is really one of contamination rather than of instream flows. Admittedly, this will not always be an easy decision, because some streams suffer from chronic problems in both categories. Water quality models used most frequently in IFIM were designed to determine the assimilative capacity of streams in wasteload allocation studies. IFIM does not routinely employ models designed to accommodate the more esoteric kinetics of heavy metals or organic pesticides. The rationale for not including these variables routinely in IFIM is that if the stream is severely polluted with a persistent pesticide, you probably have a contaminant problem first and an instream flow problem second. Engaging in an IFIM study in a seriously polluted stream might divert resources that could be better allocated to alleviating the contamination. Moreover, changing the flow regime in a seriously contaminated stream may not do any good anyway.

Data Requirements

Inputs to water quality models are similar to models used solely for water temperature; they simply have more components: water quality constituents, definition of stream network, hydraulics (flow, velocity, depth), forcing functions (loads), oxygen demand coefficients (bio-oxidation, settling/scour, reaeration, sediment oxygen demand [SOD]), nutrient coefficients (nitrification rates, nutrient-algal interactions and hydrolysis, sediment), algal coefficients (growth, respiration, saturation, light extinction), and meteorology (including evaporation).

Sources of Data

Good water quality data are more difficult to obtain than good temperature data. Rarely are daily water quality data available as maximum and minimum values, much less on a continuous basis. Information on water quality often consists of grab samples collected over a relatively brief period for a specific purpose, often with analysis techniques or reporting units that may not be standard from one application to the next. Nonetheless, careful probing can often reveal a sufficient quantity of data to aid in drawing

conclusions in a reconnaissance-level study or generating boundary and baseline conditions for a modeling assessment. Sources of data include those listed for water temperature with the addition of the more specialized databases maintained by the principals of the National Water Data Exchange (NAWDEX), namely USGS's National Water Data Storage and Retrieval System (WATSTORE) and USEPA's Storage and Retrieval System (STORET). Both of these databases are now, or will soon be, available on CD-ROM from commercial sources. As more and more data appear on the Internet, flow and water temperature will become easier to find there. As of this writing, the Internet address for NAWDEX is http://h2o.er.usgs.gov/public/nawdex/nawdex.html.

Because of the spotty nature of the data, it is a good idea to make sure that you get all the data available regardless of reporting units or analysis methods in order to develop robust relations with flow and time of year. In particular, if either stream sediment loading or reservoir modeling are your principle objectives, pay the most attention to data values that accompany the relatively rare high flows that often contribute the largest physical loadings. A number of esoteric techniques have been developed to extrapolate constituent values for high flows. We have found practical advice given by Ferguson (1986). For the intrepid stickler for detail, however, Cohn et al. (1989) offers state-of-the-art discussion of techniques.

Microhabitat Analysis

The models of the microhabitat simulation component of IFIM are known collectively as the Physical Habitat Simulation System, or PHABSIM (Milhous et al. 1989). PHABSIM consists of three parts: (1) channel structure, (2) hydraulic simulation, and (3) habitat suitability criteria (Fig. 3-9). Channel structure incorporates all of the fixed channel properties that do not change dynamically with streamflow (although they may change gradually over long time periods). Examples of fixed channel characteristics include the dimensions and cross-sectional configuration of the channel, substrate characteristics and distribution, and the locations of various types of structural cover within the channel. Hydraulic variables are those that change dynamically as a function of discharge, for example, water surface elevations, depths, velocities, wetted perimeters, and surface areas. Hydraulic simulation programs are used to predict the values of these hydraulic variables at discharges that were not measured. Habitat suitability criteria (HSC) are used to define the ranges of depths and velocities, as well as what types of cover and which characteristics of the substrate are important to a species or life stage of a species.

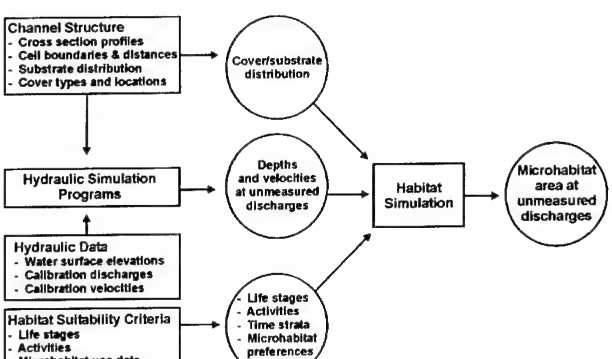

Fig. 3-9. Components and information flow in the Physical Habitat Simulation System.

Physical microhabitat data for PHABSIM are collected along transects. For the most part, the same types of data are collected on the transect regardless of the stream setting. With few exceptions, data collection is fairly routine and standardized once the transects have been established. During IFIM study planning, the real issues are how to represent the stream segments and at what level of detail.

Within a segment, there are several habitat-related subdivisions: reaches, mesohabitats, and microhabitats. A reach is typically about an order of magnitude longer than the width of the channel (commonly 10-15 channel widths), and contains many or all of the meso- and microhabitat types present in the entire segment. Mesohabitat types are roughly the same scale as the channel width and delineated by localized slope, channel shape, and structure. Riffles, runs, glides, shoals, pools, pocket waters, and divided channels are names of different kinds of mesohabitats. Microhabitats are defined as relatively homogeneous areas, about the same scale as used by an individual fish engaged in a specific activity, such as feeding or spawning. Tree-snags, undercut banks, the tail-outs of pools, mid-channel gravel bars, and velocity shelters behind boulders are all examples of channel subunits at the microhabitat scale.

Over the past 15 years, two very different strategies have evolved for the representation of a segment: representative reach characterization and mesohabitat typing. A representative reach (Bovee 1982) is approximately 10-15 channel widths in length and is assumed to contain essentially all of the mesohabitat types of the segment. The characterizing trait of the representative reach is that mesohabitat types tend to occur in a repetitive pattern. This concept was derived from Leopold et al. (1964), who noted that riffles and crossing bars in alluvial streams tended to be spaced longitudinally at a distance equal to about 5-7 times the channel width. The reasoning behind the representative reach was that each major mesohabitat type should be represented at least once in a relatively long reach of stream. Because of the repetitive nature of alluvial channels, it was also reasoned that the proportions

of the segment's mesohabitat types could be represented in a single reach. Because the uniformity of spacing observed by Leopold et al. (1964) applies only to alluvial channels, however, it follows that the criteria for a representative reach will be best met in an alluvial stream.

Selecting a representative reach is very straightforward and may not even require a trip to the field. The basic tool is a USGS 7.5 min topographic map (or several maps taped together) to encompass the entire segment. Lengths of stream equivalent to 10 to 15 channel widths are marked off from the bottom to the top of the segment and numbered sequentially. For example, if the stream channel is 100 m wide, reaches of 1,000-1,500 m would be marked off on the map. Unrepresentative (e.g., bridge crossings) and inaccessible (e.g., landowner permission cannot be obtained) reaches would then be eliminated from the pool. Several candidates would be picked at random from the remaining population, and the candidate reaches visited. If all the candidate reaches look fairly similar, two or more representative reaches having the best access would be selected. If all the candidates look considerably different from one another, investigators usually forego the use of representative reaches and move instead to mesohabitat typing.

In mesohabitat typing (also known as habitat mapping), as developed by Morhardt et al. (1983), the mesohabitat becomes the unit of stratification. The principles of mesohabitat typing are:

1) Mesohabitat types are defined for the stream under investigation.
2) An on-site inventory is conducted to determine the proportion of the segment represented by each mesohabitat type.
3) Two or more mesohabitat reaches representing each type are selected at random.
4) Transects are established to represent the mesohabitat type.
5) Transects in each mesohabitat type are weighted according to the proportion of the mesohabitat type in the segment.
6) The segment is represented by all of the transects from all of the mesohabitat types, combined into a single data set.

When choosing a representation strategy, it is wise to consider the characteristics of both the technique and the locale to which it will be applied. In different streams, one technique may be better suited than the other, and in some places, the representation strategy has been institutionalized (e.g., habitat mapping is the preferred approach in California). The representative reach approach is most appropriate in channels where habitat types tend to occur in repetitive sequences. Usually, this means that representative reaches are more applicable to alluvial streams, but some colluvial channels also exhibit a considerable amount of repetition in the distribution of habitat types. Mesohabitat typing is more appropriate in streams that exhibit a more random distribution in habitat types or large variability in their sizes. An assumption of the representative reach approach is that the reach contains all of the mesohabitats present in the segment, in approximately the same proportion as the segment. Experience has shown, however, that representative reaches do not always contain all of the mesohabitat types that are present in the segment, nor do they always contain mesohabitat types in the correct proportions. In theory, mesohabitat typing bypasses these problems. Beginning users of this technique have discovered, however, that identifying habitat types consistently is extremely difficult, especially if there is a large turnover in the members of the group conducting the stream inventories. Regardless of the representation approach, collecting PHABSIM data in replicate reaches or mesohabitat types is recommended (although not mandatory) to improve the accuracy of the microhabitat estimates. Therefore, one important decision the study planning team must make is which approach to use in which segments and how many replicates to measure.

Data Requirements

PHABSIM can be used with remarkably little data. The quality and credibility of the output, however, is enhanced by collecting a critical mass of information for each of PHABSIM's components:

1) Channel structure - required data include distances between transects, dimensions of stream cells, and channel geometry data (paired measurements of distances and streambed elevations across the channel). Optional but highly recommended data are descriptions of the substrate composition and cover types at each location (called a vertical) where channel geometry data were collected.
2) Hydraulic data - required data include the measurement of water surface elevations and the corresponding discharge at each cross-section. Under steady flow conditions, the discharge only needs to be measured at one cross-section. We highly recommend, however, the measurement of a calibration set of mean column velocities and two additional pairs of water surface elevations and discharges. In the overall scheme of things, the low and high calibration discharges should differ by at least an order of magnitude. Additional water surface elevation and discharge data are advisable in sites exhibiting complex hydraulics (e.g., pocket waters or high gradient reaches) or divided channels (e.g., islands or braided). Other optional hydraulic data include the measurement of nose (usually benthic) velocities and additional sets of mean column velocities. Many biologists believe that near-bottom velocities provide a more accurate representation of microhabitat conditions for benthic organisms. Additional sets of mean column velocities may be

useful as supplementary calibration data or for quality assurance assessments of model performance.

3) Habitat suitability criteria - required data are habitat suitability criteria for the species of interest. These criteria may be obtained from literature sources or expert opinion or empirically developed on-site. For criteria that are not developed on-site, an evaluation of their transferability to the stream under study (called the destination stream) should be conducted. We recommend field testing criteria empirically, if possible, especially in high-profile or controversial studies.

Sources of Data

The type of physical microhabitat data needed for a PHABSIM analysis will usually not be available for the stream under study. Even if cross-section or hydraulic data are available from another source, it is unlikely that it will be collected in the right place or in a manner appropriate for a detailed microhabitat analysis. For example, the most widely available cross-section data exist at USGS gaging stations. These cross-sections are commonly located at bridge crossings, however, which make poor representations of the microhabitat available in a stream. As a result, site-specific physical microhabitat data nearly always fall into the category of study implementation. It is not so much a question of whether these data will need to be collected, but rather, where, how much, and in what amount of detail.

Habitat suitability criteria present a number of options during the study planning phase of IFIM. The good news is that habitat suitability criteria are available from a wide range of sources, including the USGS Biological Resources Division. The bad news is that the existence of habitat suitability criteria does not guarantee their applicability to a particular IFIM study. Criteria can be presented in different formats, they may be incomplete, or their accuracy may be questionable. The variables used to define the criteria might not represent the variables considered to be most important to the target species. Therefore, it may be necessary to develop habitat suitability criteria on-site, or test their transferability during the implementation phase.

Schedules and Budgets

Two of our most common technical assistance questions are "How much does it cost to conduct an IFIM study?" and "How long does it take?" Our typical answer for both questions is "It depends." The time and cost needed to complete a study are directly related to the work to be done: that is, to the geographical extent and complexity of the study area, how much information needs to be "back-filled" to analyze alternatives, the number of study sites where data are to be collected, the logistics involved with data collection, complexity of model calibration and synthesis, difficulty of identifying and evaluating viable alternatives, and the setting in which problem resolution will take place.

Regardless of the complexity of the study, the scope of work must always be reconciled with project deadlines and budgets. If the study plan cannot be implemented within the constraints of time and resources:

1) the geographical extent of the study area may be reduced to concentrate on the most seriously impacted areas,
2) the number of measurement sites, replicates, and the level of detail may be reduced,
3) the amount of calibration data may be reduced,
4) some quality control standards may be relaxed,
5) the budget may be increased to allow more personnel and equipment to be brought on-line (in cooperative efforts, stakeholders often pool their resources to increase efficiency), and
6) deadlines may be extended.

Salaries and travel expenses usually represent the largest dollar costs for conducting IFIM studies. The cost of conducting a study is directly related to the time involved in data collection, model calibration and quality assurance, synthesis of model results, preparation and evaluation of alternatives, and negotiating a settlement. Therefore, the time required to perform all the various tasks can be reasonably converted into a study budget. Conversely, study planners confronted with a predetermined budget can work backwards using time estimates to determine the scope and complexity that could reasonably be expected.

There are actually two aspects of time estimates that are relevant to study planning. First is the elapsed time to completion (ETC) for a particular activity, which is highly influenced by seasonal factors. The ETC is important for establishing realistic deadlines and milestones but is not very useful in appraising costs. To arrive at a good estimate of cost, it is also necessary to determine the actual people-time involved in each portion of the study.

Scheduling Field Work

For some of IFIM's component models, data collection is a low-intensity, year-round effort. Other types of data collection are punctuated by flurries of activity, driven by biological or hydrologic factors. Water temperature and meteorological data are examples of data that can be collected with data loggers, providing relatively easy year-round data coverage. Temperature may be of greater concern during summer, however, so other data related to temperature modeling (e.g., stream shade parameters) might be focused on summer conditions. In most applications of IFIM, the greatest expense involves the collection of data for PHABSIM.

Activities associated with developing or testing habitat suitability criteria will be dictated primarily by biological factors but to some extent by hydrology as well. It is impossible to develop or test spawning criteria, for example, when the target species is not spawning. Likewise, it may

not be feasible to collect information on habitat use at high flows for reasons of safety, poor visibility, or gear limitations.

In alluvial channels, PHABSIM data should be collected on either the rising limb or falling limb of the hydrograph. It is wise to avoid straddling the peak of the hydrograph in this case, because if the channel structure is going to be altered by high discharges, this is when it will happen. In northern climates, ice breakup may also affect channel structure, so it may be advisable to avoid straddling the winter period. Channel changes are less likely in bedrock or colluvial streams, so scheduling field work around the hydrologic regime is a lesser concern in these streams.

A significant planning decision involves the number of calibration velocities to be measured for PHABSIM simulations. The collection of velocity data can be very time-consuming and costly, representing a potentially serious trade-off between cost and accuracy. It is fairly common practice in noncomplex streams and noncontroversial studies to collect only one set of velocity calibration data. In this case, we recommend that velocities be measured at a moderately high discharge, so calibration velocities are available for a large portion of the active channel. In complex channels, multiple sets of calibration velocities may be needed to avoid large errors in velocity prediction. If a second velocity set is warranted, it may be better to collect the data at a relatively low flow, owing to the increased complexity of flow patterns around rocks and other obstructions. It may also be necessary to collect velocity data at other discharges for model verification, especially if the results of the study are headed for litigation.

Water surface elevations should also be measured at several discharges. Normally, three sets of water surface elevation data are recommended, although five or more may be needed in divided, braided, or other complex channels. Calibration sets should correspond to the flows at which calibration velocities were measured, but they should also be measured at a sufficiently wide range to simulate all of the discharges in the hydrologic baseline. Milhous et al. (1989) provide a rule-of-thumb to help determine what the calibration flows ought to be. They suggest an extrapolation range for PHABSIM simulations of 0.4 times the lowest calibration flow to 2.5 times the highest. Thus, to determine the lowest and highest calibration flows, one needs only to find the lowest and highest discharges in the hydrologic baseline and divide by 0.4 and 2.5, respectively. The models can be calibrated with less separation in calibration discharges, but more expertise in hydraulic modeling will be needed to perform the calibration.

Alluvial streams. Although PHABSIM field work is usually conducted during summer, the guidelines presented above suggest a slightly different approach to scheduling in alluvial streams. Inventory-type activities can be conducted in early spring, before runoff, or in early- to mid-summer, as soon as the water clears up enough to distinguish habitat types. Spring spawning activities may start in March or April, so development or testing of habitat suitability criteria might also be initiated at this time. Unless all of the data can be collected on the rising hydrograph, measurements of channel profile and hydrographic data should be deferred until runoff begins to subside. The recessional limb of the hydrograph is a good time to measure high- and mid-flow water surface elevations, as well as calibration velocities. August and September are usually ideal times for conducting surveys of the cross-section profiles, collecting substrate and cover data, and obtaining low-flow water surface elevations. Many fall-spawners will be active during October and November, a good time to develop or test their habitat suitability criteria. Normally, there is little reason to be out in the stream during winter, unless habitat suitability criteria are being developed or tested. A typical work schedule for an IFIM study is illustrated in Fig. 3-10.

Nonalluvial streams. In colluvial or bedrock channels, sites can be established and channel geometry data can be collected nearly any time. Because the streambed is more visible at low-flow, it may be desirable to obtain channel geometry data (especially substrate and cover) sometime between late summer and early spring. (Note: Surveying is easier during late fall because it is easier to obtain a clear line-of-sight when there are no leaves on the trees.) At higher flows, investigators should concentrate on measuring several widely separated water surface elevation-discharge pairs at all sites in the study area. Calibration velocities and any remaining water surface profiles can then be measured leisurely on the falling limb of the hydrograph.

Estimating Time Requirements

Hydrology component. The time required to assemble a hydrologic baseline is negligible when there are concurrent USGS gaging station records for all of the segments in the study area. Some time is required to move existing

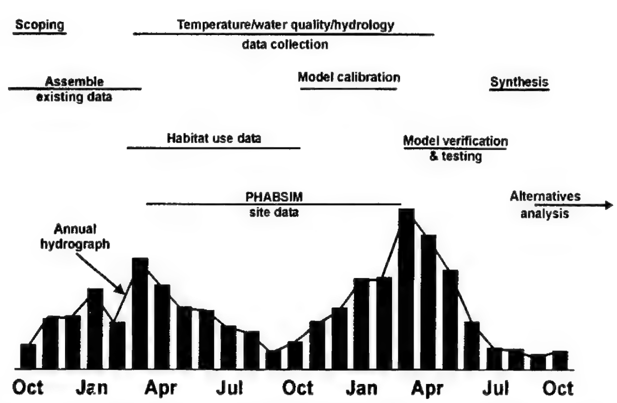

Fig. 3-10. Typical scheduling of an IFIM project. The duration of a project can vary from 1-10 years, depending on complexity and geographic extent.

hydrologic data from their source to the files that will be used in developing and analyzing alternatives, however. It will take several hours to several days (depending on the period of record and the time-step) to rekey hydrologic data that were retrieved on hard-copy. A fraction of that time is needed if the data are retrieved from CD-ROM or from the USGS website.

Time requirements for the hydrologic components become more significant when the baseline must be synthesized. If the synthesis can be achieved by the extension of records, it will normally take 3-5 days to develop the regression models and perform error analyses for each short-record segment. Because this type of hydrograph synthesis is based on existing data, it can be performed any time during the study. Time becomes a serious matter when hydrographs must be synthesized in ungaged streams. The elapsed time to assemble a good set of paired data for the semipermanent and long-term gages may take 3-9 months, depending on the seasonality and variability of streamflow.

Many applications of IFIM involve reservoir operations models, particularly for the formulation and evaluation of streamflow-related alternatives. For site-specific studies or linear networks, relatively simple, mass-balance models can be programmed in a day or two. These generic models will usually perform adequately to address most of the issues related to simple linear networks. If the problem involves a parallel or compound hydrologic network, however, it will be necessary to employ a real reservoir operations model, complete with flow routing capabilities (the model incorporates travel times as well as volumes). If the water management entity in control of the reservoir has already developed a reservoir operations model, there may be little effect on the total time requirement (the ETC) to analyze alternatives. If there is no reservoir model or the one in existence must be modified in order to be useful in alternatives analysis, however, it may take 6 months to a year to bring the model on-line.

Temperature. Many of the data collection activities associated with temperature modeling involve low-intensity monitoring, especially for air and water temperatures. The advent of the digital thermograph has made the handling and manipulation of large volumes of temperature data essentially trivial. Other data related to temperature modeling can be obtained from different components of IFIM or can be collected simultaneously with data for other components. For example, channel geometry data can be synthesized from PHABSIM measurements and hydrology data from the hydrologic component. The only other site-specific data of real significance are related to stream shading components, and these measurements can be made concurrently with PHABSIM data collection. Because stream shading can vary seasonally, it may be necessary to repeat the measurements when foliage is present and absent. If summer temperatures are of greatest concern, however, it may suffice to collect stream shading data only for the summer months.

The real-time commitment for temperature modeling occurs during model calibration and quality assurance evaluations. One of the characteristics of temperature modeling is that it takes place in a highly interlinked simulation environment. What happens in one segment is transmitted to the next, so it is common practice to simultaneously calibrate at all of the verification nodes, rather than calibrating them one at a time. This approach is actually the most efficient way to calibrate the temperature model, but the interdependencies within the modeling network can make calibration difficult and frustrating at times. Depending on the complexity of the network, calibration and quality assurance related to the temperature component will usually take a skilled person somewhere between 2-4 weeks to complete. Unfortunately, calibrating temperature models is one of those activities where adding people to the task does not necessarily help get the job done faster.

Water quality. What pertains to temperature modeling with respect to data collection and model calibration also applies to water quality modeling. As mentioned previously, the water quality constituents that are normally included in an IFIM application do not require sophisticated chemical analysis equipment, such as mass spectrometers or gas chromatographs. In recent years, advancements in multiparameter data logging equipment has made it possible to monitor common water quality constituents as easily as temperature. The difference in price between a temperature data logger and one for water quality modeling, however, is likely to place some severe limits on the number of instruments that can be deployed.

Several important chemical parameters cannot be measured directly in the field. Foremost among these is BOD, which requires a sample to be incubated under specified conditions for a period of time. The loading rate (i.e., the volume and concentration) of BOD is continuously monitored from point sources such as municipal water treatment plants. These data can usually be obtained from the plant manager or from regulatory agencies. Background BOD (i.e., that entering the stream from headwater areas) or BOD contributions from nonpoint sources are seldom measured, let alone monitored. Therefore, at least some field sampling will be needed to ascertain loading rates from other than point sources along the stream. Where nonpoint sources are substantial contributors of BOD, obtaining data on BOD concentrations along the stream can be a fairly intensive activity. It is also an activity that may have significant seasonal variation, requiring more than a few grab samples during a particular season. Changes in BOD loading (plus its effects on dissolved oxygen concentration and relationship with nutrient cycling) may require year-round sampling for at least a year. The actual time involved in taking and analyzing samples will probably

not amount to more than a month, but the ETC will likely be extended a full year.

Calibrating a water quality model is similar to calibrating a temperature model, except there is more to it. For one thing, temperature is a major driving variable in water quality models, necessitating that the water temperature component be calibrated and verified first. The time required to complete model calibration, sensitivity analysis, and other forms of quality control will typically range from 1-2 weeks for site-specific applications or simple linear networks. It may take several months of full-time work to calibrate and verify water quality models in parallel or compound networks.

PHABSIM. For most applications of IFIM, PHABSIM analysis represents the single largest expenditure of time and resources. Numerous factors, some controllable and some uncontrollable, contribute to high variability in the time and expense of conducting a PHABSIM analysis. The ease or difficulty with which habitat suitability criteria can be developed or tested, the number of measurement sites and replicates in each segment, the number of transects used to describe each site, the size and complexity of the river, and logistics of getting to and moving around in the river all relate to the time and expense involved in collecting data for PHABSIM.

A bell-shaped relationship exists between the ease of data collection and the time and effort involved in developing or testing habitat suitability criteria. If the target species are all relatively abundant and the stream is easy to sample, empirical development or testing of criteria can take as little as a week or two. (Note: Developing criteria for several species at once takes little more time and effort than it does for a single species.) At the other end of the spectrum, it may only take a few days to evaluate habitat suitability criteria if it is impossible to collect habitat use data, because this situation mandates a round-table discussion among stakeholders and species experts. The intermediate scenario, where collecting empirical habitat use data is difficult but not impossible, can add up to a serious time investment if study planners are not careful. Although we advise people to conduct empirical tests of the criteria as a routine part of an IFIM study, we also counsel study planners to leave themselves a way out if transferability testing becomes disproportionate to other aspects of the study. It should be legitimate to incorporate a contingency clause in the study plan that allows for a nonempirical evaluation if the criteria cannot be tested after 1-2 months of sampling.

The geographic coverage of PHABSIM sites, the amount of replication, and the level of detail are the most controllable factors in regulating time and cost estimates. These factors collectively distill down to the number of transects that will be used to represent the segment. Although there is no fixed formula to determine the exact number of transects required for every mesohabitat type, the average number of transects used to describe single-channel mesohabitat sites usually ranges from two to six (two for the most uniform mesohabitats and five or six for the most complex). This estimate is based on our reviews of many PHABSIM studies conducted over the past two decades, including our own. However, the estimate does not include replicate measurements of each mesohabitat type, nor does it apply to sites containing multiple habitat types (representative reaches). Issues related to geographic coverage, number of replicates, and transect density can often be addressed by staging a field trip for the stakeholders. The purpose of the field trip should be to obtain consensus regarding the approximate number of transects needed in each site for planning purposes. If transect placement is new to the study planning team, it may be helpful to retain the services of an independent consulting firm or government agency with experience in PHABSIM studies to explain how they would lay out each site.

The number of transects that must be measured is only part of the story in determining the time and resource investment required for PHABSIM data collection. The amount of time required to measure each transect is equally important in the time equation. The factors affecting time-on-transect are less controllable than choices related to the number of transects used to describe the segment. We have found that the amount of crew-time expended per transect is fairly consistent, however, varying by about 50% for three streams surveyed by our staff in recent years (Table 3-2). Person-days per transect included all activities associated with PHABSIM measurements, not just the time to measure variables on individual transects.

Table 3-2 suggests that two person-days per transect is a conservative estimate of the amount of time required to collect all the data for the channel and hydraulic components of PHABSIM. The estimate can be adjusted upward or downward to reflect the perceived difficulties presented by the study sites. Somewhat surprisingly, stream size had relatively little influence on the time-on-transect estimates. Sites on large rivers tend to be longer and require more time for site layout and preparation than sites on small rivers. Large rivers, however, also allow the use of motorboats, a more efficient mode of transportation than wading (provided that portaging can be minimized). Table 3-2 also suggests that the factors determining time-on-transect are highly related to logistical challenges presented by the site. Moving around takes longer in sites where the water is deep and fast, with tricky footing or hazardous boating conditions. Logistical problems are worst in sites that alternate between being too deep to wade and too shallow to traverse in a boat.

A major determinant of the amount of time spent laying out and preparing a site for PHABSIM measurements is how difficult the site is to survey. Difficulty, in this case, is

Table 3-2. Estimates of person-days per transect required to conduct all activities related to PHABSIM data collection at 12 contrasting sites in three rivers.

River	Site	Number of transects	Channel width (m)	Moving hazards index[a]	Line-of sight index[b]	Person-days to complete	Person-days per transect
Tallapoosa	Woodland	35	40	2	2	28	0.8
Tallapoosa	Woodland†	35	75	4	2	27	0.8
Tallapoosa	Daviston†	28	120	3	2	26	0.9
Huron	Bell Road	53	35	2	5	61	1.2
Huron	Mast Road	54	45	3	6	68	1.3
Huron	Hudson Mills	52	45	3	4	60	1.2
Cache la Poudre	Riffle	11	20	1	1	8.5	0.8
Cache la Poudre	Pool 4	10	20	3	1	8.5	0.9
Cache la Poudre	Pool 3	17	20	5	1	15.5	0.9
Cache la Poudre	Pool 2†	24	20	10	2	24.5	1.0
Cache la Poudre	Pocket water	20	30	8	3	24.5	1.2
Cache la Poudre	Cascade	22	20	9	2	21.5	1.0

[a]A subjective scale from 1-10 indicating the difficulty of moving around in the site, with 1 being easiest and 10 being most difficult.
[b]A subjective scale from 1-10 indicating the difficulty of surveying the site, with 1 being easiest and 10 being most difficult.
†Extensive use of boats required at these sites.

determined by how often the surveying instrument must be moved. In Table 3-2, this factor is expressed as the line-of-sight index, because the need to move the instrument is directly related to the length of a clear line-of-sight. The line-of-sight index was rather low for the Tallapoosa River sites, which were vegetated nearly as heavily as the Huron River sites. Because an electronic surveying device was used on the Tallapoosa River, however, many of the problems associated with surveying in dense vegetation were avoided. In the Tallapoosa River, we were able to set up the instrument in mid-channel and measure elevations at distances exceeding 500 m. These capabilities were impossible with the surveying instrument used in the Huron River, where it was also too deep to set up the instrument in the middle of the channel. In contrast, the Cache la Poudre sites were so short that all the measurements could be made from a single instrument setting.

The time required to perform calibration, quality assurance, and microhabitat simulation tasks associated with PHABSIM is highly variable. In about 25% of the applications of PHABSIM, calibration and simulation is trivial; it can take longer to construct the input data files than it does to calibrate and verify the models. More typically, however, it will usually take from 1-2 days to 1-2 weeks per site to complete this portion of a PHABSIM analysis. The time required to calibrate the hydraulic models hinges on the quality of the data and the complexity of the site. The axiom "pay now or pay later" is particularly germane to PHABSIM, because inadequacies in the field data often come back to haunt the investigator in the form of more time and greater expertise required to perform high quality simulations.

Budgeting Tips

When estimating the cost of a study, do not forget to add planning time and the time involved in alternatives analysis and negotiations.

To determine the cost of field work, try to estimate the number of crew-days in the field per trip, the number of people on a crew, and the total number of trips. Estimate the cost of a crew-trip by factoring in salaries and benefits, travel expenses, and operating expenses. Allow for down-time due to weather or breakdowns (equipment and otherwise). Depending on geographical location and logistics, down-time can add up to 50% or more to the total time to conduct field work.

Keep the safari factor as low as possible. Determine the optimal crew-size for each job and try to staff your crews with the right expertise and the right number of people.

There is no substitute for experience, especially during model calibration and simulation of alternatives. Experienced modelers may cost more per hour, but they will save you money in1 the long run. Adding more people to this task reduces efficiency as often as it improves it.

Chapter 4. IFIM Phase III:

Study Implementation

Under the subject of study implementation, we describe briefly how IFIM's component models work and what it takes to make them go: data collection, model calibration, error analysis, and synthesis. This chapter is not a step-by-step manual in data collection or model calibration, but rather it summarizes what happens during the various stages of study implementation. In this chapter, we also discuss quality assurance as it applies to each of the component models of IFIM.

To be successful, the implementation phase should accomplish two goals. First is communication. If nothing else, IFIM is designed to facilitate communication and increase understanding. The second goal is to foster trust and confidence among participants and stakeholders. The implementation phase may be your best opportunity during an IFIM project to accomplish this goal. People who have worked together in at least some aspects of study implementation generally trust one another more than they do people who have been only passively involved or absent entirely.

Active participants also tend to have more faith in the procedures, data, and results from a study. Under the best of circumstances, true cooperation and teamwork will be achieved, individuals will have first-hand experience with the processes, and everyone will have had a voice in decisions made along the way. If this level of teamwork can be achieved, the resolution of the problem is likely to be more cooperative and certainly more pleasant to participants. Even if participants do not entirely trust each other, increased ownership of the processes will usually translate into greater confidence in the results.

A common predicament encountered in most environmental studies, including IFIM, is that information used in decision-making may be incomplete or unreliable. How this uncertainty is dealt with is one of the most important aspects of an IFIM analysis. Analysts may be reluctant to discuss uncertainty because it may imply a less-than-perfect application of the methodology. Such reluctance may be exacerbated by decision-makers who cannot deal with uncertainty because they believe that science is exact (Reckhow and Chapra 1983). Studies that do not acknowledge error may be considered by decision-makers to be more credible than those that do, when just the opposite is often true. This paradox often becomes most apparent during litigation, where analysts are loath to discuss their own uncertainty but eager to discuss that of their opponents.

In our previous discussion of objectives, mention was made of performance criteria. For each of the components of the methodology, performance criteria are defined by acceptable procedures and allowable errors. Quality assurance is especially important during IFIM implementation for a number of reasons. A discussion of uncertainty will educate decision-makers and clarify their expectations with respect to acceptable standards of error. Quality assurance measures help investigators determine the reliability of information that will be used to formulate and evaluate alternatives. The modular construction of IFIM enhances the importance of error analysis, because the reliability of a component (e.g., microhabitat) is often judged by the performance of its parts.

In the past, decision-makers have dealt with uncertainty by promoting instream flow recommendations that were "conservative on the side of the fish." The engineering analogy to this practice is overdesigning a project to the point that the risk of failure is almost zero. The main drawback of overdesign, however, is that the project may cost so much that it becomes financially infeasible. Overly conservative recommendations can have the same effect on instream flow studies, but the costs manifest themselves as constraints on the feasibility of alternatives.

Owing to the significance of uncertainty in IFIM studies, a major focus of this chapter will be on quality assurance and error analysis. Although we will discuss data collection and calibration procedures in general, separate manuals are available that provide the blow-by-blow details of making IFIM models go (see suggested readings). This chapter is heavily weighted toward the various quality assurance measures that should be performed during data collection, calibration, and error analysis. We will also describe a variety of analytical techniques for quantifying uncertainty. Common error standards will be discussed where appropriate. However, the standards of acceptable error are really performance criteria and should be established by the implementors and stakeholders of a study.

Hydrology

Model Concepts

Two types of spatial problems commonly occur in developing hydrologic time series. Either there are no hydrologic records for parts of the study area or existing records do not cover the same periods of record. In either case, it is likely that all or a portion of the hydrologic record will need to be synthesized for one or more segments. Hydrograph synthesis will also be necessary in the process of formulating and testing alternatives.

Two techniques are especially useful in the synthesis of baseline hydrographs for IFIM analyses: mass balancing and station regression. A third technique, using watershed models, can be employed if it is impossible to collect the data needed for the other two methods. Watershed models are less accurate than mass balancing and station regression,

however, and are typically employed only when the cost of collecting data outweighs the need for accuracy.

Mass Balancing

Mass balancing is easy and straightforward, usually a simple exercise in arithmetic. Discharges for coincidental time-steps are added or subtracted to determine the streamflow above or below the confluence of two or more gaged streams. Fig. 4-1 illustrates how mass balancing could be used to fill in missing hydrologic records in a stream gaging network. The hydrologic record for segment A can be calculated by adding together the discharges from gages 1 and 2. Streamflows in segment B can be determined by subtracting the streamflow in segment A from the discharge measured at gage 3. Using similar combinations of addition and subtraction, the records for segments C and D can also be synthesized.

Mass balancing can become more complicated if the travel time between gaging stations is larger than the time-step for the hydrologic time series. When the travel time is larger than the time-step, it becomes necessary to incorporate a time lag into the discharge calculation. For example, the discharge in segment B (Fig. 4-1) might be calculated as the current day's discharge at gage 3, minus the previous day's discharge at segment A. Time-lag corrections can be tricky because the lag interval changes as a function of discharge, being shorter at high flows and longer at low flows. Yet another advantage of using long time-steps is that the averaging period generally exceeds the travel time between measurement locations, thus eliminating the need for any special corrections.

Station Regression

The station regression approach involves the development of a model relating the streamflow records at one station to those at another and is commonly used when all the records are not concurrent. Suppose, for example, that the baseline period of record for a study is defined as 1954–94. As shown in Fig. 4-2, this period of record is available at only two of the gages in the drainage. All of the gages, however, have overlapping records from 1965–76.

The model used to extend the records for one of the short-term gages is developed by selecting discharges from coincidental time-steps from both the short- and long-term gages. The discharges are transformed into logarithms and a linear regression performed on the log-transformed data. The resulting regression model takes the form:

$$\log Q_s = \alpha + \beta \log Q_l \qquad (7)$$

where Q_s is the discharge at the short-record gage, Q_l is the discharge at the long-term gage, and α and β are regression coefficients.

In its most elementary form, station regression is a relatively simple technique for hydrograph synthesis. As with mass balancing, though, care must be taken to adjust for time lags when short time-steps are used, and additional precautions should be taken when conducting station regressions. An important consideration is that the hydrologic regimes should be very similar among all the stations used in the regression(s). Greater accuracy can be achieved if the watersheds from which the data originated have similar characteristics of elevation, aspect and orientation, prevailing meteorology, and land use patterns. The pattern of

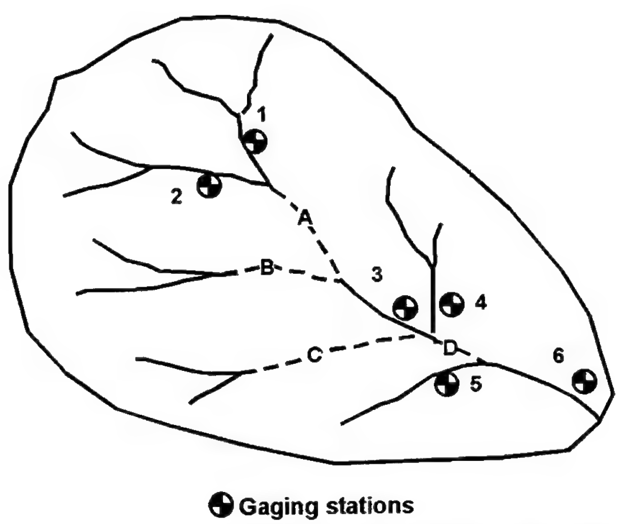

Fig. 4-1. Example of a stream gaging network where mass balancing can be used to fill in missing hydrologic data. Dashed lines indicate segments in study area where the arrangement of gaging stations allows hydrograph synthesis by mass balancing.

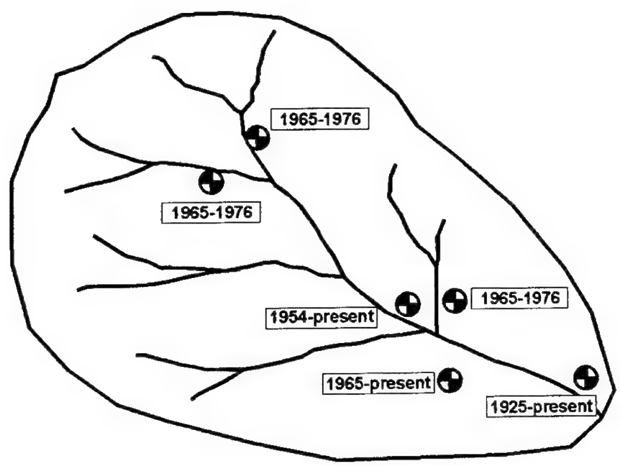

Fig. 4-2. Water supply and habitat network showing the periods of record for the gaging stations in each segment.

snowmelt of north-facing slopes, for example, is not the same as that of south-facing slopes. Differences in rainfall occur in streams draining windward and leeward sides of a mountain range. When dissimilar meteorological or runoff processes have occurred between the long- and short-term stations, the first indication is usually a large amount of scatter around the regression line and a relatively low correlation coefficient.

Hydrograph Synthesis in Ungaged Streams

A modified version of a station regression can be performed for an ungaged stream if there is at least one long-term gage (termed a reference gage) somewhere in the vicinity. Station regressions in ungaged streams differ from extensions of records in gaged streams in two significant aspects. First, it is necessary to develop a short-term hydrologic record in the ungaged stream. This is done by establishing and calibrating a semipermanent gage. Second, discharge estimates obtained from a semipermanent gage may be instantaneous, rather than averaged over some time period. Time-lag effects can be especially troublesome with instantaneous discharge estimates.

Historically, the term "semipermanent gage" has meant a staff gage (essentially a ruler attached to a fence post or bridge piling and immersed in the river). However, pressure sensors, sonic rangers, and data loggers can also be installed as semipermanent gages to obtain a continuous record. Although somewhat more expensive (approximately $1,000 per station), continuous stage recorders are recommended where streamflow is highly variable or there is a considerable travel time between the semipermanent gage and the reference gage.

If the semipermanent gage consists solely of a staff gage, station regressions must be performed using instantaneous discharges. Instantaneous flows at the temporary and long-term gages may not relate very well if runoff patterns between the two gages are desynchronized appreciably. Although there are ways to account for unsteady flow and differences in travel time, none of them are as satisfactory as using streamflows that were measured under steady conditions. The advantage of continuous stage recorders is that they permit calculation of an average streamflow over a specified time period. As the averaging interval increases, the measurements better approximate steady flow conditions.

Whether consisting of a simple staff gage or a pressure sensor/data logger, installation of a semipermanent gage follows the same guidelines as establishment of a permanent gaging station, that is:

1) gages should be conveniently located, easy to read, and protected from vandalism;
2) the elevation of the gage, relative to a known datum, should be established so the gage is recoverable if disturbed or destroyed; and
3) measures should be taken to dampen out oscillations in stage readings that result from wave action.

The most convenient locations for semipermanent gaging stations are often at bridge crossings or other places where the stream comes close to a road. Unfortunately, the more convenient the location, the greater the potential risk of vandalism. Staff gages, because of their exposure and visibility, are especially vulnerable to disturbance. Although it is possible to protect staff gages by hiding them or bolting them onto bridge pilings, it is far more important to be able to recover the gage if it is disturbed. Gages are made recoverable by surveying the elevation of the top of the gage when it is installed. If the gage is disturbed, a new one can be put in its place at exactly the same elevation as the original. This procedure ensures that all of the gage readings from the new gage will correspond with those from the old gage.

The anticipated range of river stages should also be considered during installation of the gage. Staff gages or pressure sensors that are above water at low discharges are not very useful. Similarly, it is difficult to obtain a good high flow reading from a staff gage that is completely submerged. To avoid low flow problems, the measuring device should be installed upstream from a strong hydraulic control, a feature in the channel, such as the crest of a riffle, that creates a backwater effect in an upstream direction. The gage should be placed at an elevation equal to or slightly less than the lowest elevation, or thalweg, of the control. By using a very long gage, or several short ones staggered up the bank, gage readings can be made at high flows. If several short gages are used, it is important to tie them all to a common reference elevation or datum. Normal high flows are usually no problem with pressure sensors or sonar rangers, unless the flows are so high they destroy the instruments.

The gage is rated by measuring the discharge and the staff gage reading at several widely separated discharges. These paired data are then used to develop a rating curve between the gage readings and the discharges (Fig. 4-3). With a calibrated rating curve, an investigator can immediately determine the discharge from any reading on the gage. Rating curves tend to be curvilinear when plotted on arithmetic graph paper, so it is common practice to linearize the relationship by logarithmic transformation of the stage-discharge pairs (Fig. 4-3). A simple linear regression is performed between the logarithms of the stage and the logarithms of the discharge. To find the discharge for any particular gage reading, the logarithm of the stage is found, the regression equation solved for the logarithm of the discharge, and the discharge determined from the anti-log.

Calibration

IFIM's synthesized hydrographs are "self-calibrated," because a regression model represents the best fit of the

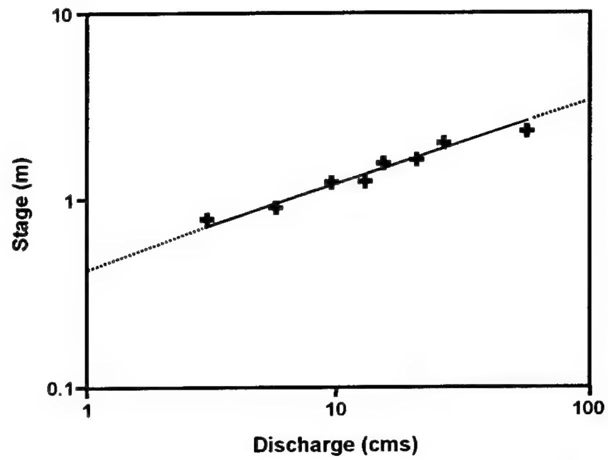

Fig. 4-3. A stage-discharge relationship, linearized by plotting data on a logarithmic scale.

data that were used to construct it. Nonetheless, an investigator may still need to choose the best overall model among several possibilities. Factors that can improve the accuracy of an empirical hydrologic model include:

1) Use of "steady flow" data. With the exception of projects involving hydropeaking or extremely flashy streams, it is usually advisable to use time-steps of several days or longer whenever possible. Using longer time-steps will help reduce errors associated with differential travel time and lag effects.
2) Comparable watershed data. The more similar two watersheds are with respect to drainage area, elevation, aspect, slope, vegetation, and land use, the more similar their runoff characteristics.
3) Use of a large, extensive database. The influences of measurement errors and between-station differences are magnified if only a few data pairs are used to develop the regression model. These effects tend to be diluted if many (e.g., 25 or more) data pairs are used to develop the model. Also, interpolation between calibration data is more accurate than extrapolation beyond the end-points. The farther apart the end-points, the more discharges can be found by interpolation.

Error Analysis

One of the first indicators of the overall quality of a station regression model is a goodness-of-fit criterion, usually expressed as a correlation coefficient (Reckhow and Chapra 1983). However, correlation coefficients are fairly coarse screening tools for synthesized hydrographs. Because the synthesized discharges will play a central role in subsequent decision-making, merely obtaining a statistically significant correlation may not be good enough. The cutoff point for accepting or rejecting a model should be decided by the stakeholder group, but alternative models should probably be considered if r^2 is less than 0.8 or so. It might be possible to improve accuracy by developing independent regression models for each month or season. Another technique, useful in drainages subject to unpredictable and localized precipitation events, is to develop a multiple-station model. These models are based on a multiple regression, using long-term data from two or more reference gages. Unfortunately, neither of these techniques will help very much in flashy drainages subject to localized thunderstorms. Here, a longer time-step may be needed to smooth out much of the variance causing the scatter in the regression. This solution is less than satisfying if the extreme events are of concern, but it may be a compromise the analyst will have to live with.

Another way to investigate model accuracy is to develop a histogram of percentage errors, also known as an error dispersion plot (Fig. 4-4). Two methods are commonly used with hydrologic data to quantify error dispersion: split sampling and the jackknife method (Mosteller and Tukey 1977). Split sampling involves dividing the total database into a calibration subset and a confirmation subset (Reckhow and Chapra 1983). The calibration subset is used to build the regression model, which is then used to predict the discharge at the temporary gage for time intervals contained in the confirmation subset. Under the jackknife method, with n data pairs, n-1 pairs are used to construct the regression model and 1 pair is used to calculate the error. The "n-1 to calibrate, 1 to confirm" procedure is repeated n times, so that each case is used once as the confirming case (Mosteller and Tukey 1977). According to Reckhow and Chapra (1983), the jackknife method is superior in instances where there is only a limited amount of data available for model construction and testing.

Prediction errors are calculated as:

$$E = \frac{(Q_p - Q_m)}{Q_m} x\ 100\% \quad \textbf{(8)}$$

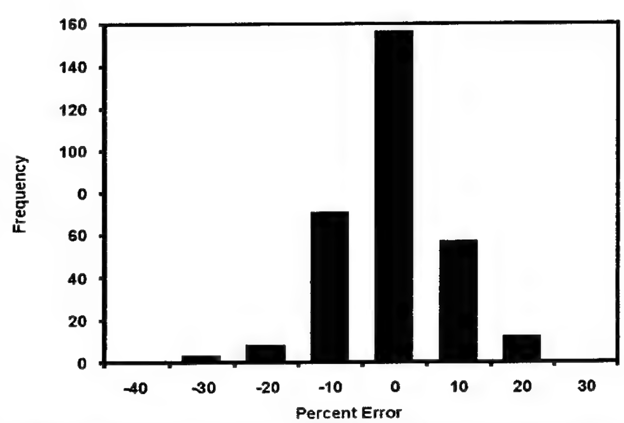

Fig. 4-4. Example of an error dispersion plot using percent errors. The heights of the bars indicate the number of predictions falling within a particular error range.

where E is the percent error, Q_p is the predicted discharge, and Q_m is the measured discharge.

Fig. 4-4 illustrates an idealized error distribution. Bartholow (1989) offered several guidelines for evaluating an error distribution that can be applied to hydrologic models. Errors should be normally distributed, unbiased and unskewed, with a mean and modal error of zero. A large proportion of the errors should be clustered around zero (i.e., between ±10%) and there should be relatively few errors in excess of 50%.

Bias can be examined by plotting the residual or percent error versus the predicted discharges. Ideally, the magnitude of the error should not change as a function of discharge. If errors become larger as the predicted discharge increases (the typical case) but the errors are equitably distributed between positive and negative, the error distribution is said to be heteroscedastic (Fig. 4-5). The error distribution lacks independence if the error is correlated with the predicted discharge (Fig. 4-6). From a practical standpoint, some heteroscedasticity is almost inevitable when prediction errors are calculated as residuals. When percent error is plotted against predicted discharge, however, this phenomenon should not be evident. Error distributions that are correlated with the predicted discharge may be symptomatic of a systematic error in the model. Investigators would be well advised to reexamine their data collection procedures and any possibilities of error introduced by variable time-lag problems between stations. At any rate, a model lacking independence in its error distribution should not be used to synthesize the baseline hydrologic time series for the segment.

Box and whisker plots (Fig. 4-7; Tukey 1977; McGill et al. 1978; Reckhow et al. 1990) are based on order statistics, much like those used to construct flow duration curves. These diagrams display information on the sample median, dispersion, skew, relative size of the data set, and statistical significance of the median. A box and whisker plot is constructed as follows:

1) Data are ordered from lowest to highest.
2) The lowest and highest values are plotted on the graph as short horizontal lines. These marks delineate the extremes, or "whisker" portions of the plot.
3) The upper and lower quartiles, analogous to the 25% and 75% exceedance values on a duration curve, are determined for the data set. These values define the upper and lower edges of the box. The median is marked with a dashed line.
4) The whiskers are drawn in between the top and bottom of the box and the corresponding maximum and minimum values.
5) The width of the box is scaled so that it represents the sample size.
6) The statistical significance of the median is indicated by the height of the notch in the box. Following McGill et al. (1978), the height of the notch above and below the median is approximated by:

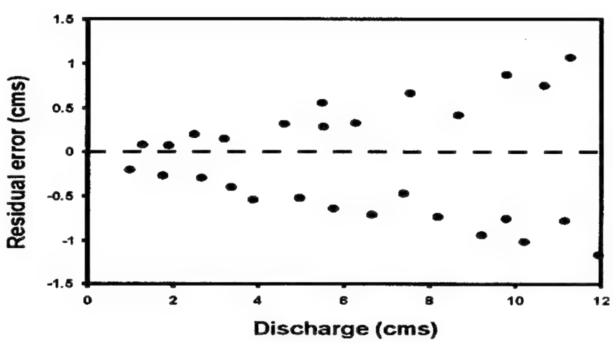

Fig. 4-5. Example of a heteroscedastic error distribution. Residuals increase with increased magnitude of the predicted discharge, but negative and positive errors are approximately balanced.

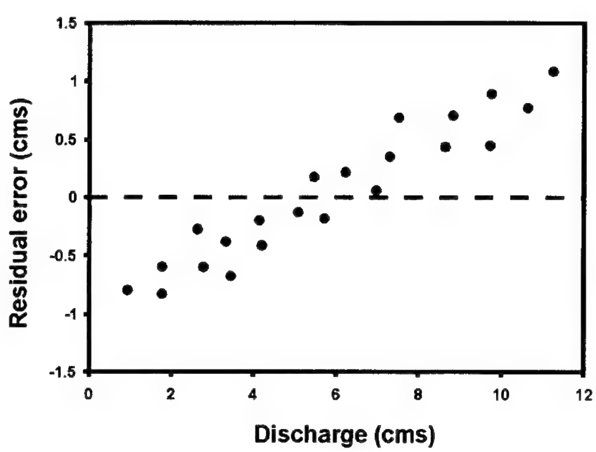

Fig. 4-6. Error distribution lacking independence. Such distributions may indicate the presence of a systematic source of error.

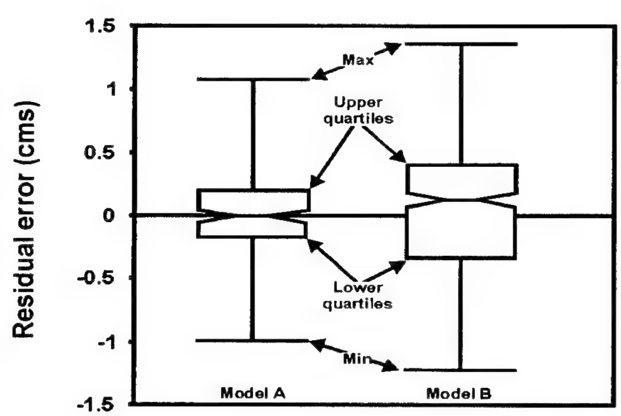

Fig. 4-7. Box and whisker plots comparing the residuals from two different hydrograph synthesis models from the Huron River, Michigan. Model A is superior to Model B because (a) its median error is zero, (b) its interquartile range is smaller, and (c) the total range of error is smaller.

$$Notch\ Limits = Median \pm \frac{1.57I}{\sqrt{n}} \qquad (9)$$

where I is the interquartile range (upper quartile minus lower quartile values) and n is the sample size. The notch is an approximation of the 95% confidence interval for the comparison of box medians. When the notches for two boxes are aligned vertically, the medians are not significantly different at about the 5% level.

Channel Geomorphology

Channel geomorphology is an important input to the temperature, water quality, and microhabitat components of IFIM. As discussed previously, channel dimensions and shape data can be measured as direct input to these models when no change in the existing channel configuration is anticipated. When a channel change must be incorporated into the IFIM analysis, however, this component can become very complex and difficult. In fact, we strongly recommend that you acquire the services of a specialist in channel dynamics if you are confronted with this problem. What follows is a brief description of the process that such a specialist might use to determine the shape, pattern, and dimensions of a new channel following a change.

Model Concepts

Two separate analyses are needed in order to incorporate a channel change model into IFIM. First, the dimensions of the new channel must be estimated. Second, the shape and pattern of mesohabitat types in the new channel must be approximated. As part of this exercise, the distribution of cover and substrate distribution in the new channel would also be determined.

Estimating Channel Dimensions

One of several channel dynamics models could be used to determine the dimensions (e.g., channel width, mean bankfull depth, hydraulic gradient) of a new equilibrium channel. Of these, the HEC-6 model, developed by the U.S. Army Corps of Engineers (1991) is perhaps the best-known and most widely used in the United States. The organization and information flow through HEC-6 is illustrated in Fig. 4-8. Channel geometry data are collected along transects to describe the structure and shape of the channel, as well as the particle size distribution of the materials making up the streambed. Hydraulic data, notably water surface elevations and discharges, are inputted to a hydraulic simulation model to predict water surface elevations and other hydraulic variables for a range of simulated discharges. A variety of computational algorithms is available to the investigator to calculate sediment transport rates in suspension as well as in bedload.

HEC-6 operates according to time-steps in a series. At the first time-step, a discharge is retrieved from the hydrology component. Sediment inflow data is obtained for the time-step from an empirical sediment-discharge curve

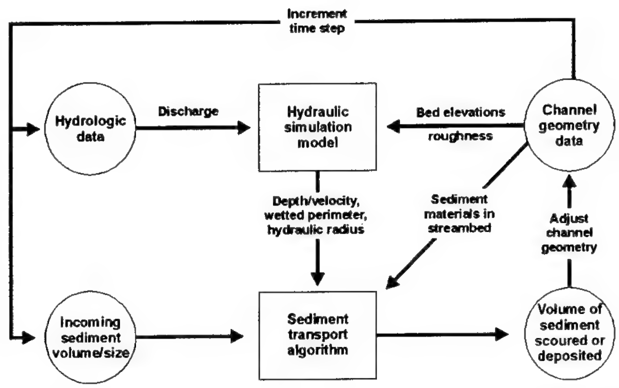

Fig. 4-8. Flow chart of the components and information flow of the HEC-6 model.

derived for the stream under study. The hydraulic simulation component produces information relating to the sediment transport capacity of the stream at the discharge for the time-step. Hydraulic characteristics, sediment inflow, and the particle size distribution of the streambed are all input variables to the sediment transport component. Users can choose from about a dozen different transport algorithms in HEC-6 to calculate the volume of sediment that will be scoured or deposited at a transect with each time-step in the series.

When scour or deposition occurs after a time-step, the bed elevation is adjusted and the change in sediment load routed to the next transect downstream. The time-step is then incremented and HEC-6 is provided with a new discharge, new sediment inflow parameters, and an adjusted channel geometry. The process is repeated until the supply of time-steps has been exhausted. This end point represents the average channel dimensions at the end of the time series. If the time series is long enough to allow the channel characteristics to stabilize in the model, HEC-6 provides an estimate of the new equilibrium condition.

Output from HEC-6 can be used directly in a one-dimensional temperature or water quality model because only average channel dimensions are needed. HEC-6 will not provide the kind of detailed information needed for PHABSIM, however. In addition, HEC-6 does not predict watershed sediment yield, so inflowing sediment loads and particle-size gradations must be provided as input. These data are usually determined from empirically derived sediment-discharge curves for the upstream ends of the main stem, tributaries, and local inflow points. Empirical sediment-discharge curves, however, are only valid if the watershed processes are themselves in a state of equilibrium. If the proposed action will alter the rate of sediment production, input to HEC-6 will need to be provided from a sediment yield model.

Estimating Channel Shape and Pattern

It may be possible to add the detail necessary for microhabitat analysis using an analogous stream model. This approach is based on the premise that changes in channel structure can be approximated from similar systems where the same kinds of change have already occurred (Kellerhals and Church 1989; Kellerhals and Miles 1996). In this situation, HEC-6 might be used to determine the new channel dimensions, depth of aggradation or degradation, and slope under a new equilibrium condition. Detailed habitat measurements of the type used in IFIM would be made in a similar channel which has already adjusted to a similar proposed action. These measurements would then be scaled up or down to fit the dimensions predicted by HEC-6.

The scaling process itself involves two steps. First, the channel pattern and mesohabitat distributions in the analogous stream are measured and related to the bankfull channel width of the target stream, as determined from HEC-6. For example, the meander wavelength and the riffle-pool spacing in alluvial channels are both related to the bankfull width (Leopold and Maddock 1953; Leopold et al. 1964). If the target stream will be larger than its analog,the distance between meanders and the lengths of pools, riffles, and other mesohabitat features should be proportionately longer. The scaling factor will affect the distances assigned to transects measured in the analog.

The second step involves surveying cross-sections in the analog stream and scaling their bankfull dimensions up or down to fit the average bankfull dimensions of the target stream, as predicted by HEC-6. It is important during this step not to distort the shape of the channel too much during the scaling process. For this reason, it is helpful if you can select an analog stream that has about the same average dimensions as predicted for the target stream. The scaled dimensions (along with the estimated distances between transects) would then be entered into the hydraulic simulation component of PHABSIM to complete the analysis.

A common misconception is that channel dynamics are ignored in IFIM. In truth, the capability exists, but the current state-of-the-art for estimating channel shape and pattern is primitive, at best. If stakeholders are comfortable using the estimates and best guesses of an analogous stream model, there is no reason that the model output cannot be used as input to PHABSIM. Because of the choices presented by available channel change models, however, most users confronted with this problem usually choose an alternative course of action (e.g., ignore it, wait it out, or proceed anyway). In the future, as two-dimensional hydraulic models evolve for instream flow analyses, obtaining better predictions of localized scour and deposition at a relatively small scale may become possible. Such capability is bound to improve the accuracy of the channel morphology component of IFIM. Unfortunately, this technology is still in its infancy for large-scale riverine applications and may not be ready for IFIM applications for many years.

Water Temperature

Model Concepts

The stream temperature model most often associated with IFIM applications is SNTEMP (Theurer et al. 1984). SNTEMP is a mechanistic, one-dimensional heat transport model that predicts the daily mean and maximum water temperature as a function of stream distance and environmental heat flux. Net heat flux is calculated as the sum of heat to or from long-wave atmospheric radiation, direct short-wave solar radiation, topographic radiation, convection, conduction, evaporation, streambed fluid friction, and the water's back radiation (Fig. 4-9). Direct radiation is attenuated by topographic and vegetative shading, as well as by cloud cover. The heat flux model also incorporates thermal mediation from groundwater inflow.

The heat transport model is based on a dynamic temperature-steady flow equation and assumes that all input data, including meteorological and hydrological variables, can be represented by 24-h averages. SNTEMP is applicable to a stream network of any size or order. It includes (1) a solar model to predict the solar radiation penetrating the water as a function of latitude and time of year; (2) a shade model that quantifies riparian and topographic shading; (3) algorithms that correct air temperature, relative humidity, and atmospheric pressure for changes in elevation within the watershed; and (4) regression algorithms that smooth and/or fill missing observed water temperature measurements. Turbulence is assumed to thoroughly mix the stream both vertically and transversely. Timesteps ranging from 1 month to 1 day have been used in SNTEMP.

SNTEMP requires that the spatial layout of the hydrologic network be divided into segments like those described earlier. Segments used for SNTEMP, however, may be subdivisions of the segments used for habitat accounting in IFIM. In addition to having homogeneous streamflow characteristics, segments for SNTEMP are defined according to width, slope, roughness (Manning's n) or travel time, and shading characteristics. The meteorological influences are air temperature, relative humidity, wind speed, percent possible sun (inverse of cloud cover), and ground-level solar radiation. Water flow into the segment and groundwater accretions along the segment, along with their temperatures, are also required inputs.

Because SNTEMP is complex, the basic processes have been abstracted into a simplified version—the Stream Segment Temperature Model, or SSTEMP. Currently, there are three programs making up the segment family: SSTEMP for temperature modeling, SSSHADE for shade estimation, and SSSOLAR for solar radiation estimation (Bartholow,

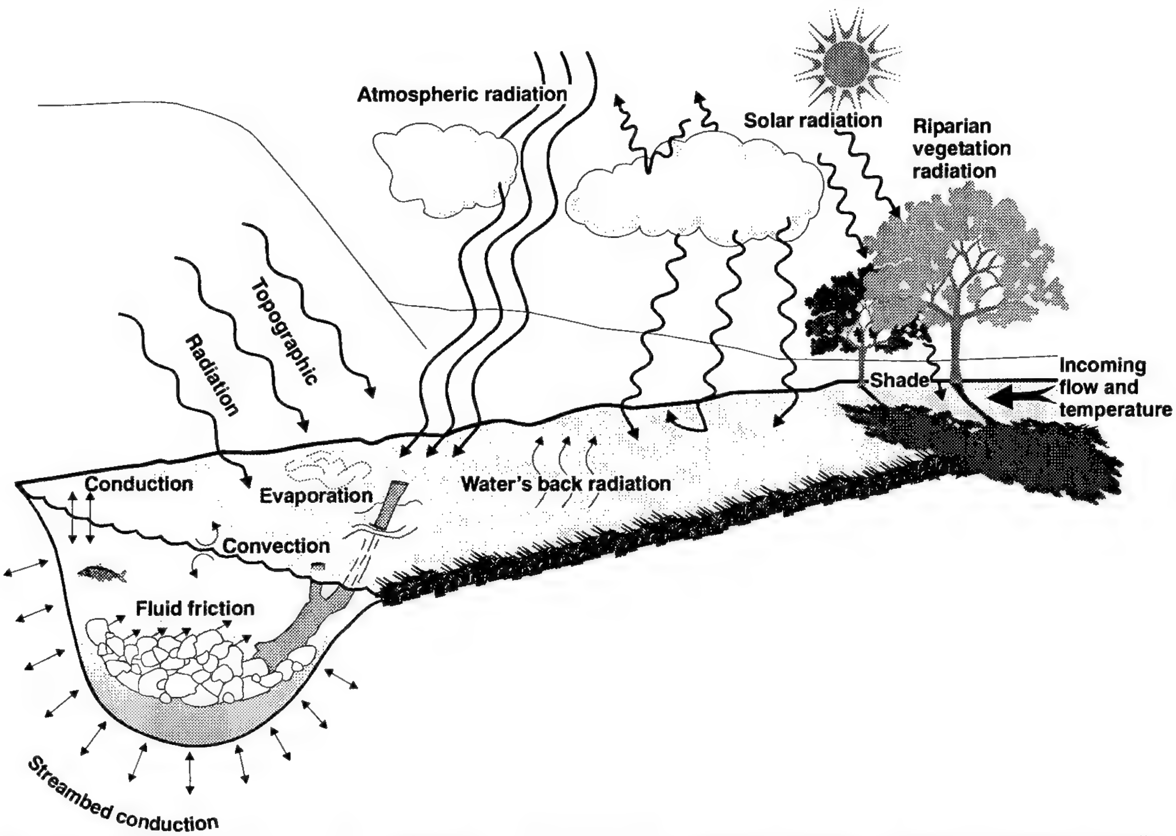

Fig. 4-9. Heat flux components used by SNTEMP to calculate the energy balance between the stream and the surrounding environment.

J.M. (unpublished); Stream Segment Temperature Model (SSTEMP) Technical Note 2; Stream Segment Shade Model (SSSHADE) Technical Note 3; and Stream Segment Solar Model (SSSOLAR) Technical Note 4. Computer programs and documentation for IBM-PC. Available from River Systems Management Section, USGS/BRD, Fort Collins, Colorado). This class of programs has proven valuable for handling a few stream reaches and simple networks for a few time periods. SSTEMP is also a friendlier way to learn about temperature modelingthan diving straight into the network model. Data input parameters may range from "back of the envelope" types of calculations to detailed micro-meteorological field measurements, with corresponding degrees of reliability. However, the SSTEMP models become tedious and error-prone as the number of stream segments or time periods increases. Nonetheless, these segment models may be used for many temperature modeling applications. One important use is as a screening tool during study planning to determine whether a more sophisticated temperature analysis is warranted.

Data Collection

Despite an impressive list of input variables for a water temperature model, only limited empirical data are collected in the field. Much of the data used in the temperature model can be "piggy-backed" with data collected for other IFIM components. For example, channel data from PHABSIM sites can serve double duty in the temperature model. Hydrologic data, with the exception of groundwater inflow, will routinely be derived for the hydrologic component and can also be used as input to the temperature model. In fact, if a very thorough job is done in putting together the stream gaging network (e.g., one or more gages are established in every segment), it may be possible to estimate groundwater inflow by mass balance from the combined records for the stations. The types of data that are most likely to be collected in the field include (approximately in this order) water temperature, stream shading, and meteorology, particularly air temperature.

Water Temperature Data

Whenever possible, water temperature data should be collected with continuously recording thermographs. Several considerations go into decisions on how many temperature recording instruments should be used and where they should be placed. The number is often based on cost. Obviously, the first priority is to accurately measure stream temperatures within reaches of biological importance. Such locations also suffice for calibration purposes. Beyond this first priority, the picture becomes cloudy, with

many intervening variables. In general, the next priority must be assigned to reservoir release temperatures, since all stream temperature models require these starting water temperatures. Sometimes, if it is known that reservoir release temperatures are relatively constant, at least through the season of concern, grab samples may be adequate for calibration purposes. If the segment furthest upstream in the study area is more than 30 km downstream from a reservoir, however, the need for release temperature measurements decreases. In such cases, equilibrium release temperatures may suffice. This is not to say that release temperature should not be measured, but if time, money, and labor are limiting, this may be an area where data can be sacrificed. If, however, the release temperature fluctuates dramatically, or more important, release temperature is a management action to be evaluated, placing a recorder at that location should be a priority.

In a situation where there is no reservoir, headwaters are logical candidates as places to monitor water temperatures. However, the temperature of headwaters more than 30 km upstream from the uppermost segment can be approximated by using the "zero flow headwater" approach. This is a shortcut approximation of the groundwater temperature, estimated as equal to the mean annual air temperature.

The confluences of major tributaries are also prime locations for water temperature measurements. For our purposes, a "major" tributary is defined more on the basis of its potential effect on temperature rather than by streamflow. For example, a tributary that changes the temperature of the mainstem by more than 1° C should be defined as "major." The mixing equation:

$$T_c = \frac{(Q_a T_a + Q_b T_b)}{(Q_a + Q_b)} \qquad \textbf{(10)}$$

where T_c is the mixed temperature below the confluence of streams *a* and *b*, Q_a and Q_b are the discharges, and T_a and T_b are the temperatures of streams *a* and *b*, respectively, may be used to estimate temperature change below the confluence of a tributary. This equation can also be used to determine if a tributary will affect mainstem temperatures under altered or postproject conditions (provided an estimate of altered tributary temperatures).

Beyond these general rules, one can only say, "The more measurement locations, the better." Greater instrumentation provides insurance against inevitable downtime and lost data and will also help isolate troublesome reaches for which the models seem to perform poorly. More monitoring stations also add to the cost, although the price of digital thermographs has fallen dramatically in recent years. In small streams (e.g., average annual discharge ≤ 2 cms), the maximum density of recorders that might be needed is probably no greater than one every 5 km. For larger rivers, one recorder every 10 km may be adequate.

Collecting water temperature data has become much easier with the arrival of continuously recording data loggers. The advent of the digital thermograph has been a tremendous advance over the analog (strip chart) recorder. Recent improvements in digital thermographs have resulted in smaller, less expensive units with expanded capabilities over similar units produced only a few years ago. Whereas downloading temperature data was once a tedious, winter-long chore, it can now be done by computer in a matter of minutes. Digital thermographs may also be more accurate than the older analog models, but whatever the type, we recommend calibrating the instrument in a water bath against an ASTM (American Society for Testing and Materials) thermometer prior to deployment.

Despite advancements in recording equipment, though, obtaining a continuous, uninterrupted set of temperature data remains a challenge. Arrayed from most to least common, problems associated with thermographs include (1) theft, (2) vandalism, (3) leakage, (4) battery failure, (5) chart jam or malfunction, (6) stylus jam or breakage, (7) RAM failure, (8) chip pin damage or breakage, (9) analog to digital converter failure, and (10) tape or film failure.

Equipment failures may be alleviated somewhat by visiting the instrument more often. Once every 2 weeks to begin with, and then once a month, may be reasonable, but this is highly dependent on local conditions, desired redundancy, and cost. You have about the same security choices with thermographs as with staff gages: either hide them or make them foolproof. As with staff gages, most experienced field practitioners prefer foolproofing. A typical form of theft or vandalism protection involves encasing the thermograph in a heavy, sealed iron pipe and attaching the casing to an immovable object with a piece of log chain or heavy cable.

Thermographs should be installed far enough downstream from thermal point sources to ensure complete mixing. The sensor itself is usually mounted in a perforated pipe directly in the streamflow but protected to minimize physical damage. The sensor should not rest in direct contact with the streambed, nor should it be in direct sunlight, if possible. Obviously, erroneous measurements will result if the sensor is exposed to air at low flows or becomes covered with silt or other debris.

Groundwater

Many streams receive substantial portions of their flow for all or part of a year from groundwater. Having a good estimate of groundwater discharge and temperature during periods of low streamflow is especially important. Commonly, the diurnal temperature fluctuation may be almost completely damped out in spring-fed streams, especially in large or heavily shaded streams (Moore 1967). Localized influx of cool groundwater may account for temperature reductions of 4-5° C over a distance of 300 m in small streams (Smith and Lavis 1975).

Stevens et al. (1975) recommend using maximum-minimum thermometers for measuring groundwater temperatures at a reconnaissance level. This seems quite practical, since groundwater would not be expected to fluctuate markedly in temperature. Other measures would need to be taken should variation be observed; temperatures may be measured in wells, springs, mine shafts, or holes bored in the stream bank. If on-site groundwater temperature measurements are unavailable, the mean annual air temperature is a good approximation (Currier and Hughes 1980; Theurer et al. 1984), except in geothermal areas (Moore 1967).

Stream Shading Data

Data that must be provided to the shade model include topographic elevation, vegetation height, crown diameter, vegetation density, and the offset of vegetation from the edge of the stream (Fig. 4-10). These data must be recorded for the east and west banks of the river, the conventions for which are described in Bartholow (1989).

Topographic elevation refers to the angle from horizontal to the topographic horizon, measured from the center of the stream. Vegetation height is usually found by trigonometry, using angle measurements made with a clinometer. Average offset and crown diameters are estimated using tape measures. Vegetation density, expressed as a percent shading parameter, can be determined as the ratio of light meter readings under the canopy and in direct sunlight. None of these measurements is particularly difficult, and the collection of shade data can easily be incorporated into the data collection activities for PHABSIM. Generally, shade data are most important during summer, but it may be necessary to collect additional data during other seasons.

Meteorological Data

One of the most serious problems with meteorological data for water temperature modeling is that long-term weather records may not be very representative of the conditions at the study area. For example, weather station data are often collected at airports, but an airport would be an odd place to conduct an instream flow study. Similar considerations are proximity to oceans or other large water bodies, topographic characteristics, and thermal inversions. In particular, we are concerned about the representativeness of air temperature and relative humidity, two of the most influential meteorological variables used in the temperature models. Both can be very different on-site than at a long-established weather station far away in a concrete jungle. If there is any question regarding the representativeness of these data, we recommend that at a minimum, air temperature and humidity measurements be taken on-site for some period of time to allow comparative calculations if necessary.

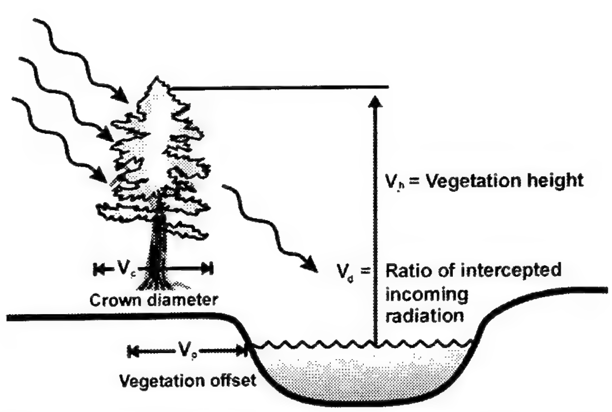

Fig. 4-10. Riparian vegetation shade parameters used as inputs to water temperature models. Vegetation density is determined as the ratio of intercepted incoming radiation.

Percent possible sun is measured at more weather stations than is solar radiation, but it is subject to more error. Technically, percent possible sun is measured as the number of minutes of direct sunlight divided by the number of minutes possible for that latitude and time of year. Obvious problems arise in determining the threshold of cloud cover at which the sun "ceases to shine." When not measured by an instrument, percent possible sun is periodically estimated by a weather observer. Estimates of cloud cover are likely to be either missing or in error at night. Since percent possible sun is used as a surrogate for cloud cover, those measurements that are taken may not be good estimates for nighttime conditions, especially in areas with marked diurnal weather patterns. None of these measurements really get at the "quality" of the cloud cover. Cirrus and nimbus clouds provide markedly different types of radiation attenuation and atmospheric reradiation. In short, percent possible sun estimates may be a good candidate for model calibration: if you have poor estimates, treat them with the uncertainty they are due.

Wind is the meteorological parameter that one would least like to translate from off-site, because the effects of topography are too varied and complex. An exception might be for biweekly or monthly time-steps. On shorter time-steps, if you cannot measure wind speed, use it as a calibration parameter. In other words, you may vary wind speed in the models within some reasonable bounds to create a better match between observed and simulated water temperatures. Some water temperature models use wind speed almost exclusively as a calibration parameter. A variety of devices are available for measuring wind speed. Unlike standard meteorological measurements, however, we are not interested in anemometer measurements from a 5-m tower. Wind speed should be measured near the water's surface, subject to the typical constraints (soil banks, riparian vegetation) at that level.

The transferability of meteorological data from the nearest measurement station to the study area is an important consideration. The bad news is that, in reality, many of the

variables measured at a meteorological station are probably not very transferable to the study area. The good news is that SNTEMP may not be sensitive to these variables. Remember that water temperature is most closely related to air temperature and the same equipment can be used to measure both. If it becomes obvious that meteorological data cannot be transferred accurately to the study area, it may be necessary to install a temporary meteorological station on-site. Data from the temporary site can be related to those from a permanent weather station using the same techniques for station regressions discussed previously.

Calibration

Recall that a temperature model consists of two basic parts: a heat flux component and a heat transport component. Each component contains parameters that affect the rates at which heat is gained, lost, and moved through the segment or network. Calibration of a temperature model involves a process of determining the "proper" values for the model's rate parameters. The basic idea of calibration is to adjust these rate parameters until the predicted water temperature agrees with the measured temperature within a set of specified standards. In reality, this procedure is repeated for many time-steps and locations until a mix of parameter values is found that gives the best overall agreement between measured and predicted temperatures.

The distinction between empirical data and calibration parameters can sometimes become a bit blurred. Input data that can be measured accurately are rarely tinkered with as calibration parameters. Data that cannot be measured very accurately or may not be very transferable are prime candidates for tweaking to fit measured temperatures. For example, air temperatures that were monitored continuously in the segment would normally be entered directly into the model with no modification. In contrast, relative humidity data collected at an airport 50 km away would be a likely calibration factor. Table 4-1 contains a listing of variables and parameters supplied to the stream segment model. The amenity of variables or parameters to adjustment for calibration purposes is illustrated by its "calibration potential."

Error Analysis

Table 4-1 illustrates one of the dangers inherent to a model with a large number of calibration parameters, the possibility of arriving at the right answer for the wrong reason. That is, the predicted and measured calibration temperatures might match fairly closely, even though one or more of the parameters was seriously in error. Only when temperatures outside the calibration ranges are simulated and compared with measured temperatures will the real effect of these calibration errors become evident.

To avoid problems with misleading calibration results, many experienced modelers advocate conducting a sensitivity analysis during the calibration process (Reckhow and Chapra 1983). A sensitivity analysis is a test of a model in which the value of a single variable or parameter is changed and the impact of the change on the dependent variable is observed. The process is conducted one variable at a time, with all other variables held constant during

Table 4-1. Variables and parameters supplied to the stream segment temperature (SSTEMP) model, with associated "calibration potentials" that indicate how likely it is that the variable or parameter will be adjusted for calibration purposes.

Variable or parameter	Calibration potential	Variable or parameter	Calibration potential
Latitude	Low	Segment length	Low
Valley orientation	Low	Manning's *n*	High
Topographic altitude	Moderate	Upstream elevation	Low
Vegetation height	Moderate	Downstream elevation	Low
Crown diameter	Moderate	Width α Term ‡	Moderate
Vegetation offset	Moderate	Width β Term ‡	Moderate
Foliage density	High	Thermal gradient	Low
Solar radiation	Low		
Air temperature	Low-moderate		
Relative humidity	Moderate	Wind speed	High
Segment inflow	Low-moderate	Possible sun	High
Inflow temperature	Low-moderate	Daylight length	Low
Segment outflow	Low-moderate	Ground temperature	Low
Lateral temperature	Low-moderate		

‡ The α and β terms of the width are hydraulic geometry coefficients relating the top width of the stream to the discharge by the equation: $w = \alpha Q^{\beta}$.

a particular trial. Using a variation of the sensitivity analysis, termed a first-order error analysis, the investigator changes the value of each parameter by a fixed percentage during each trial.

Sensitivity analysis can provide several types of useful information, both to investigators and observers of the process. The effects of errors in each of the variables and parameters on the dependent variable can be determined. This knowledge allows the investigator to identify sensitive variables (those which have a significant influence on the dependent variable) that must be reliably estimated. Conversely, sensitivity analysis also identifies variables and parameters to which the model is insensitive. Sensitivity analysis also has a practical perspective for stakeholders and observers not directly involved in the calibration process. If a sensitivity analysis was conducted, it is likely that the investigator knew what he or she was doing. Sometimes, confidence in the modeler is more important than confidence in the model.

The same types of error analyses described for hydrograph synthesis can be performed on temperature predictions. Error dispersion and bias tests should be routinely conducted on the results from a temperature model. Either a split-sampling or jackknife procedure can be used for developing the verification database. Because temperature databases tend to be rather large, however, the split-sample approach may be more practical.

Bartholow (1989) has suggested the following standards of acceptable error based on error dispersion and bias tests:

1) Mean error - The mean of the absolute values of the simulated temperatures minus the mean of the observed temperatures over all time-steps and all geographic locations should be $\leq 0.5°$ C.
2) Dispersion error - No more than 10% of the simulated temperatures should be more than 1° C from the measured temperatures.
3) Maximum error - No single simulated temperature should be more than 1.5° C from the measured temperatures.
4) There should be no trend in spatial, temporal, or prediction error.

SNTEMP and SSTEMP are capable of generating temperature data either in the form of a time-step average or as a maximum/minimum value. From a modeling standpoint, averages are probably a little more accurate and somewhat easier to calibrate than extreme values. About the only time that maximum/minimum temperatures would be considered to be absolutely essential is if the temperature criteria for the target species are based on short-term survival values or if temperature interacts synergistically with water quality factors. Where the temperature data will be used to determine directive, controlling, or growth-related factors, time-step averages will probably suffice. In contrast, evidence that survival rates or reproductive success are related to growth rates would suggest that degree-day information should be generated in addition to normal temperature model output.

Water temperature output should be obtained at numerous locations, called output nodes, along the stream. Temperature output is usually provided at each output node for a specific set of discharge and meteorological conditions. By plotting temperature against distance, you can generate a temperature profile for the segment (or multiple segments) for each set of input conditions. Look at the shape of the temperature profile and your temperature suitability curves. If the temperature is changing rapidly within a sensitive range of the target species, output nodes should be close together (i.e., about a kilometer apart). If the temperature changes slowly or the area of rapid change is outside the sensitive range for the target species, a wider spacing of output nodes is acceptable. It is a small matter to insert additional output nodes. It is not a small matter, however, to redo the entire analysis because the original output nodes were too far apart.

Water Quality

Almost any water quality model could be used in conjunction with an IFIM analysis. If there is a "standard" water quality model in IFIM, however, it is probably QUAL-2E. This program was developed cooperatively by the National Council for Air and Stream Improvement, the Department of Civil Engineering at Tufts University, and the U.S. Environmental Protection Agency (USEPA). The USEPA's Environmental Research Laboratory in Athens, Georgia, currently maintains and distributes this well-documented program, supplies limited technical assistance, and offers irregularly scheduled training classes. What follows is a brief description of QUAL-2E and a condensed comparison with a few other widely accepted models in use today. One note that applies to all the models is that their use may be limited to ice-free conditions.

Model Concepts

QUAL-2E is considered to be the standard water quality model for small streams and medium-sized rivers (Brown and Barnwell 1987). QUAL-2E simulates up to 15 water quality constituents, including temperature, dissolved oxygen, nitrogen (organic, ammonia, nitrite, nitrate), phosphorous (organic and dissolved), algae as Chlorophyll *a*, biochemical oxygen demand (ultimate or 5-day), up to three conservative minerals, and coliform bacteria. The program handles a generalized dendritic stream network with tributaries and junctions, but there are some constraints on the number of nodes (locations in the network where inputs and outputs may occur). It accepts multiple external loads and point discharges, nonpoint sources and sinks, unsimulated tributaries, and water withdrawals. QUAL-2E uses contemporary modeling theory, which handles quasi-steady-state hydraulics. Though not as sophisticated as

the dynamic flow models, QUAL-2E is academically recognized as incorporating state-of-the-art diurnal kinetics, especially for algal-nutrient interactions and the complete nitrogen series. Thus, dynamic water quality may be explored if the user supplies diurnal meteorological data. The simplicity of this one-dimensional, steady-state model means that it is relatively easy to calibrate and validate. QUAL-2E has two additional strengths. First, it lets you compute the amount of flow augmentation necessary to reach a specified dissolved oxygen goal at a given geographic location. Second, it operates in either English or International units. Some data entry "helper" programs are available, and an interesting risk analysis package for modeling under uncertainty is also available as the model QUAL-2EU. Using QUAL-2E for comparison, Table 4-2 illustrates some alternative water quality models.

QUAL-2E is the model of choice for most applications in streams and rivers, as it is designed explicitly for this purpose. It is extremely well documented and is thoroughly recognized as the standard by academic and industry professionals through repeated applications. QUAL-2E has been polished to be user-friendly in terms of input and output options and does not take a tremendous amount of computer literacy to use. The program is in the public domain and supported by a Federal agency with (at least occasional) training and technical assistance. Computer resources to use the model are reasonable. Special cases of water quality modeling, however, may require more sophisticated approaches than QUAL-2E can handle. Undoubtedly, many water quality problems will involve storage reservoir operations and quality concerns. In this case, WASP or HEC-5Q should be used. If ice conditions are involved, it may be necessary to contact someone at the U.S. Army Corps of Engineers Cold Regions Environmental Lab in Hanover, New Hampshire (603-646-4100) for assistance. For more information on water quality modeling or ecological responses, contact the U.S. Environmental Protection Agency, Environmental Research Laboratory, Athens, Georgia, 30613.

Data Collection

One of the first steps in implementing a water quality model is to establish sampling locations to be used as calibration or verification nodes for the model. Some general characteristics of a water quality sampling network are summarized in Fig. 4-11, including (1) the upstream or headwater of each stream segment to be modeled, (2) mouths of all significant tributaries (> 5-10% of flow or mass loadings) not otherwise included explicitly in the model, (3) effluent samples for all point sources before they enter the stream, (4) upstream and downstream ends of segments affected by nonpoint sources, and (5) the downstream end of the study area.

As with the temperature model, the upstream extremities are needed to define boundary conditions of flows and background concentrations. The tributary and effluent data are needed to define the loading rates. Nonpoint source data can be problematic because you often must assume negligible water quality changes in the main channel to calculate the nonpoint contribution. As in the temperature model, data collected at the downstream end is largely for

Table 4-2. Comparison of alternative water quality models that could be used instead of QUAL-2E in an application of the IFIM.

Program name	Source	Description
STEADY	U.S. Army Corps of Engineers Waterways Experiment Station (USACE-WES)	One-dimensional, steady-state, first-order kinetics only (no algal dynamics). Simple and easy to use. User documentation available.
BLTM	USGS	Reaction kinetics comparable to QUAL-2E but more flexible user specification of those kinetics. Differs in substantial ways from QUAL-2E in the advection and dispersion components of the model.
CE-QUAL-RIV1	USACE-WES	One-dimensional, dynamic flow hydraulic and water quality model used in peaking studies. Highly sophisticated. More difficult to learn and to calibrate than most other water quality models. Considered to be developmental. Support infrastructure is weak.
HEC-5Q	USACE Hydrologic Engineering Center (USACE-HEC)	Relatively simplistic water quality model, piggybacked on a sophisticated water management program (HEC-5). It only handles temperature, DO, and three conservative and three nonconservative constituents. Well-documented and supported, but no formal training provided.
WASP-4	USEPA (Athens, Georgia)	Comparable to QUAL-2E but handles dynamic flows, toxics, and eutrophication. More complex data collection and calibration than QUAL-2E.

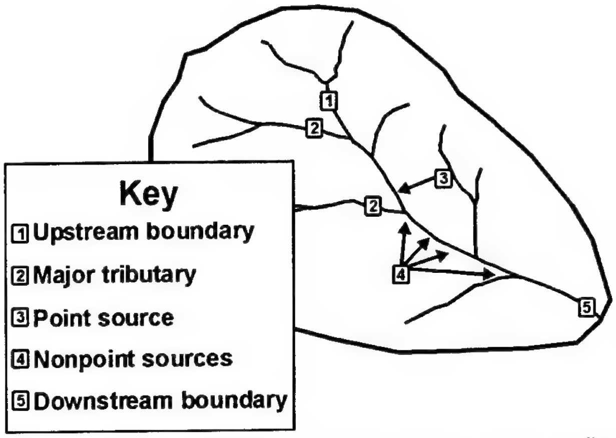

Fig. 4-11. Recommended sampling locations for water quality constituents in a stream network.

calibration and validation. If more resources are available, additional stations may be established in biologically sensitive areas, locations where known or suspected water quality violations may be occurring, and areas where stream geometry changes are likely to cause kinetic changes. All intermediate locations add additional discriminitive power and help to assure that you are not getting the right results for the wrong reasons.

Sampling frequency for water temperature is commonly done hourly as digital recorders have become so cost effective. From the hourly data, minimum, maximum, and average temperatures may be easily calculated. This may or may not be true for other water quality constituents depending on instruments available. Ideally, all measurements should be taken for suspected worst-case situations. In addition, measurements should be taken for other meteorological and flow conditions to improve the accuracy of the models in identifying incremental improvements from discharge, and to improve the statistical power of validation tests if required.

For protocols for sampling and analyzing water quality constituents, we refer the reader to American Public Health Association (1995), which covers the sampling, treatment, and analysis of the full range of water quality variables. The protocols specified in this volume are standard for water quality analysis, and stakeholders will expect these methods to be followed.

As discussed under Phase II, we are primarily interested in organic decomposition and the dissolved oxygen cycle in water quality applications in IFIM. Water quality models require essentially the same data as the hydrologic and water temperature models, plus background concentrations and loading rates, uptake, and temperature-modified reaction rates. Sampling sites are typically static and are chosen to provide representative measures of the concentration or parameter of interest. For waste assimilation studies, constituent concentrations are typically measured continuously or are grab-sampled at intervals over a 24-h period. Fixed sampling sites may be better suited for situations in which the loading rates vary temporally (U.S. Environmental Protection Agency 1985). Sampling duration should be at least one travel time through the system, and the clock time of sampling any station should be staggered to account for travel time between stations. A variation of staggered sampling is to introduce a dye-tracer and then float downstream at the same rate as the marked slug of water, sampling continuously or at intervals. This technique is preferred where the travel time through the system is several days or where loadings vary more over distance (e.g., multiple nonpoint loadings) than they do over time.

Regardless of the sampling strategy, the important constituents to measure for an oxygen-balance study include (roughly in descending order of importance):

1) dissolved oxygen concentration,
2) temperature,
3) biochemical oxygen demand,
4) discharge (river and point sources),
5) ammonia, nitrite, and nitrate concentrations,
6) sediment oxygen demand,
7) chlorophyll *a*,
8) phosphate concentrations, and
9) light.

An oxygen-balance model contains an impressive list of reaction coefficients for which values must be provided in preparation for calibration (Table 4-3). Values for many of these coefficients can be obtained from literature sources. However, most of these coefficients are provided as a range of values, which means they can legitimately be adjusted upward or downward during the calibration process.

In addition to information relating to calibration and error analysis of water quality models used with IFIM, the implementors should also document sources of information used to develop the habitat suitability indexes for water quality. Common sources of such information are from State or Federal water quality standards or from bioassay studies. Generally, the results from bioassays will be the most conservative, State standards the next most, and Federal standards the least restrictive. In selecting a particular set of macrohabitat criteria, you should check two characteristics: how the criteria were derived and what life stages and species they were derived for. If the macrohabitat criteria were based on minimum lethal dose (e.g., the concentration required to kill a single member of the target species) you may consider the criteria to be conservative. The most conservative criterion, of course, is one known to result in optimum growth rates and zero mortality. Be suspicious of criteria based on the LC-50, as these will allow at least 50% of the organisms to die before the red flag goes up. It is also important to note the species and life stages used to formulate the criteria, especially when State or Federal

Table 4-3. Typical reaction coefficients and sources of information for the QUAL-2E oxygen balance model. Sources were either literature (Lit.) or laboratory (Lab.) tests. Generally speaking, literature sources tend to be for reaction coefficients that exhibit relative consistency from study to study. Laboratory sources indicate potential site-specific variability.

Variable	Source	Variable	Source
Ratio chlorophyll α to algae	Lit.	Benthic NH_3 source	Lab.
Fraction algae as nitrogen	Lit.	Organic N settling rate	Lit.
Fraction algae as phosphorus	Lit.	Organic P settling rate	Lit.
O_2 produced per unit algae growth	Lab.	Nonconservative settling rate	Lit.
O_2 consumed per unit algae respired	Lab.	Benthic nonconservative source rate	Lit.
O_2 consumed by oxidized NH_3	Lit.	Carbonaceous deoxygenation rate	Lit.
O_2 consumed by oxidized NO_2	Lit.	Reaeration rate	Lit.
Maximum algal growth rate	Lab.	BOD settling rate	Lit.
Algal respiration rate	Lab.	Sediment oxygen demand	Lit.
Half-saturation constant for light	Lit.	Coliform die-off rate	Lit.
Half-saturation constant for N	Lit.	Nonconservative decay rate	Lit.
Half-saturation constant for P	Lit.	Biological oxidation, NH_3 to NO_2	Lit.
Nonalgal light extinction coefficient	Lit.	Biological oxidation, NO_2 to NO_3	Lit.
Algal self-shading coefficient	Lit.	Hydrolysis of organic N to NH_3	Lit.
Algal NH_3 preference factor	Lit.	Decay of organic P to dissolved P	Lit.
Benthic dissolved P source	Lit.		

standards are used. These standards are often based on general groups of species. It is not uncommon to perform bioassays on fathead minnows to formulate water quality criteria for warmwater streams. These criteria will be more conservative than criteria based on goldfish, but perhaps less so than criteria determined for rainbow darters.

At the State level, it is common for the State's department of health to formulate water quality standards, sometimes with input from the State fish and wildlife agency. At the Federal level, the USEPA has primary responsibility for assembling and promulgating water quality standards. These standards are published by the USEPA (sometimes called the EPA red book, blue book, or gold book) and contain good descriptions of the methods and rationale for individual standards. If it is unclear exactly how the standards were formulated, contact the appropriate agency for details.

Calibration and Error Analysis

Water quality models rarely lend themselves to the tight calibrations attainable with temperature models. Generally, the calibration focuses on getting the model prediction to fall within the observed longitudinal variation of the various water quality parameters. In this sense, there is not much distinction between calibration and error analysis, as the former is generally accomplished via the latter. "Eyeball" or "good-enough" statements typically are employed because the internal model calibration parameters are so uncertain. The numerous variables and adjustable rate parameters (e.g., Table 4-3) in a water quality model make getting the right answer for the wrong reason nearly as easy as getting the wrong answer. The model may perform well to describe the existing situation but be totally in error when simulating water quality under some altered regime.

Error analysis usually takes the form of (1) sensitivity analysis, (2) first-order error analysis, or (3) Monte Carlo simulation. We have already discussed sensitivity analysis and first-order error analysis as they apply to calibration of temperature models. Monte Carlo simulations expand on those ideas by specifying a statistical distribution for relevant input variables.

The person(s) who collected the data and calibrated the model will know more about the behavior of the system and the model than anyone else. Error analysis is vitally important because it can guide the investigator toward those variables and parameters that may need to be better defined or measured if the model performs poorly. If you get a blank look when you ask about error analysis, consider seeking a second opinion from a water quality modeling expert. Such expertise can usually be found at major universities in the departments of sanitary or environmental engineering, at the U.S. Army Corps of Engineers Waterways Experiment Station in Vicksburg, Mississippi, or at the USEPA lab in Athens, Georgia. If you are still early in the study implementation phase, collecting additional data or recalibrating existing data may be possible. A more difficult problem arises if concerns about error are not raised until most of the analysis has been conducted. The delay caused by the need to collect more data or recalibrate would

require much of the work to be redone, possibly at additional expense to the applicant or the consulting firm, and would require an extension of the deadline.

Two Types of Macrohabitat Output

Although this section has dealt with the theory and data requirements for component macrohabitat models of IFIM, we should not lose sight of the final output from the temperature and water quality components. The models can produce biologically relevant information, such as degree-day accumulations, at specified locations along the river (Fig 4-12). Such information is important in evaluating chronic effects of water quality or temperature: those that may affect growth rates and secondary survival, but do not cause direct mortality (Bovee et al. 1994). These effects are usually not integrated with microhabitat for time series analysis but are evaluated parallel to the habitat analysis.

The models also produce steady-state longitudinal profiles of the temperature or the concentration of water quality constituents up- and downriver. By superimposing information specifying acceptable temperatures or concentrations for the target species (macrohabitat criteria), we obtain a length of stream having suitable macrohabitat conditions at a particular discharge (Fig. 4-13). This version of macrohabitat analysis addresses the acute effects of temperature or water quality. The standards for water quality and temperature, as established by the U.S. Environmental Protection Agency or by individual states, can serve as an estimate of the acceptable limits for these macrohabitat variables.

Perhaps the most important thing to look for is whether the nonconservative constituents behave according to first-order reactions. One quick way to address this issue is to find out if the stream has an appreciable amount of algae in it. If it does, there is a strong probability that there will be second-order kinetics that cause extreme diurnal or seasonal deviations from the mean. Also, look at the

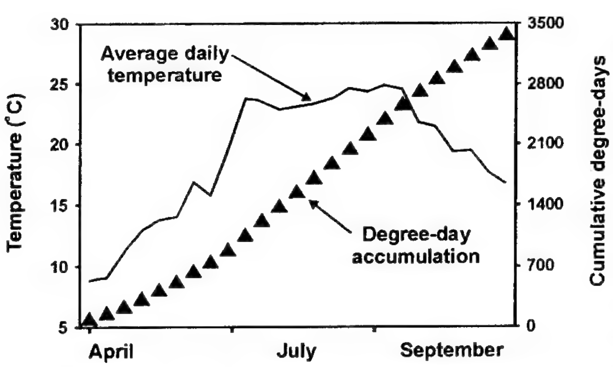

Fig. 4-12. Spatially fixed, time-variable output from a macrohabitat model: a site-specific time series of spring and summer water temperatures and accumulated degree days.

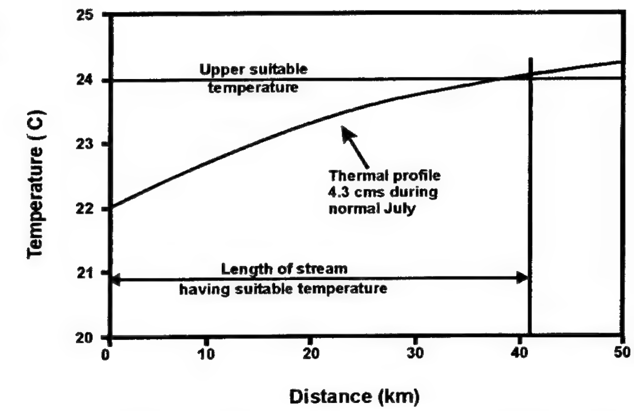

Fig. 4-13. Temporally fixed, discharge-specific, and spatially varied output from a macrohabitat model: a longitudinal temperature profile showing the length of stream having suitable temperatures at a discharge of 4.3 cms during an average July.

calibration data to see if sampling was conducted around the clock. The fluctuation of dissolved oxygen, in particular, can give you an instant idea about the severity of oxygen depletion due to respiration and may give you clues about other potential problems, such as ammonia build-up due to algal decay.

You should contact the stakeholders and other study participants if second-order kinetics are strongly implicated and not included in the model. There are two potentially serious problems here. One is that the original model used for the simulations either does not have the capability or an adequate database to perform second-order kinetic simulations. The second problem may be that the persons conducting the simulations do not have this capability. In either event, the result will probably be that data gaps will need to be identified and filled, models will need to be recalibrated or replaced, and all results based on the original simulations will need to be redone. This additional work will probably result in an increased cost to the applicant and an extension of the deadline.

Physical Microhabitat

Model Concepts

The conceptual model for PHABSIM is a depiction of the site (whether representative reach or mesohabitat type) as a mosaic of stream cells (Fig. 4-14). The lengths and widths of the cells are determined by the investigators on-site. At any particular streamflow, each stream cell has a unique combination of surface area, depth, velocity, substrate, and cover. When another discharge is simulated in the hydraulics component, the depths and velocities in all of the cells change (in cells near the edge, the surface areas may also change).

The physical mosaic provides a picture of what the stream environment looks like at each simulated streamflow. To

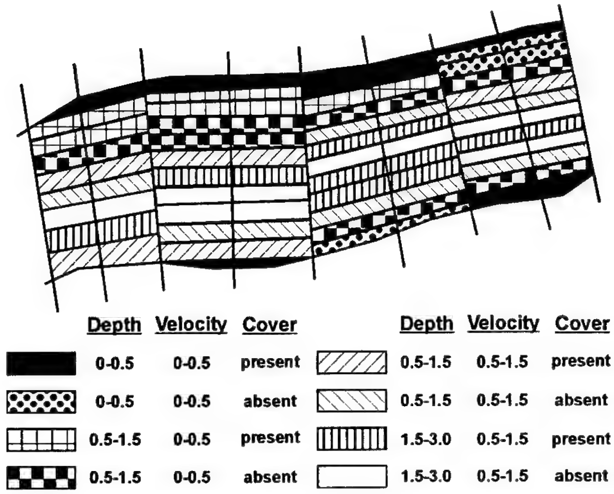

Fig. 4-14. Computer "map" of stream cells portraying the distributions of depth, velocity, and cover features from the hydraulic and channel structure models of PHABSIM.

translate this picture into an estimate of microhabitat at a particular discharge, habitat suitability criteria are used to define a suitability index for the depth, velocity, cover type, and substrate attributes of each stream cell for a life stage of a species. These univariate suitability indexes are aggregated mathematically to determine the composite suitability of the cells (Fig. 4-15), usually expressed on a scale ranging between 0 and 1. When the composite suitability is multiplied by the surface area of the cell, the product is known as weighted usable area (*WUA*). The equation is expressed as:

$$WUA_{Q,s} = \sum_{i=1}^{n} (a_{i,Q})(csi_{i,Q,s}) \qquad \textbf{(11)}$$

where $WUA_{Q,s}$ is the weighted usable area of the reach at flow (Q) for target species (s), a_i is the surface area of cell (i), and $csi_{i,Q,s}$ is the composite suitability of cell (i) at flow (Q) for target species (s). The default option is to calculate the composite suitability index as the product of the univariate suitabilities for each microhabitat variable:

$$csi = (si_d)(si_v)(si_{ci}) \qquad \textbf{(12)}$$

where si_d is the suitability index for the depth of the cell, si_v is the suitability index for the velocity of the cell, and si_{ci} is the suitability index for the channel index (usually cover or substrate) of the cell. Two other options are the geometric mean of the univariate suitabilities or the selection of the minimum suitability index as the *csi*.

Weighted usable areas (or other microhabitat metrics) are calculated for every discharge entered into the hydraulic simulation component and for every target organism selected by the user. These calculations result in the typical output from PHABSIM, a functional relationship between discharge and physical microhabitat for each target organism (Fig. 4-16). Target organisms commonly include different life stages or seasonal microhabitat for stream fishes, but microhabitat for algae, aquatic insects, crustaceans, mollusks, reptiles, amphibians, and birds have also been simulated successfully using PHABSIM. Furthermore, PHABSIM has been used to quantify the relative values of different streamflows for a variety of recreational activities ranging from kayaking to fly-fishing.

Habitat Suitability Criteria

Successful implementation of PHABSIM starts with the acquisition of accurate, realistic habitat suitability criteria for the target organism(s) being evaluated. In order to fully appreciate what the terms "accurate and realistic" can mean, it is necessary to introduce some concepts regarding the different kinds of criteria that one might use with PHABSIM. Important distinctions include the format in

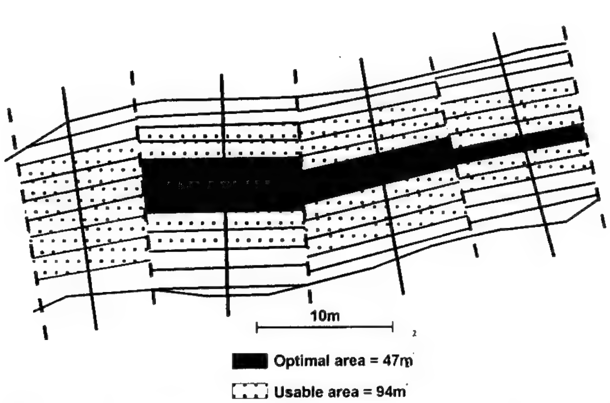

Fig. 4-15. PHABSIM's translation of the distribution of structural and hydraulic characteristics into an area of suitable microhabitat for a target species.

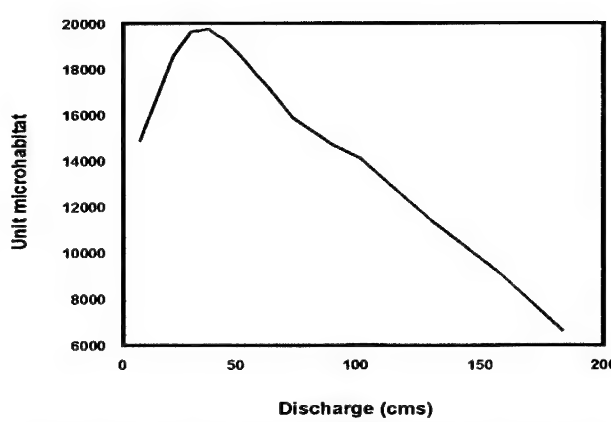

Fig. 4-16. Typical output from PHABSIM: a functional relationship between discharge and unit microhabitat area for a specific target organism. The units of microhabitat are expressed as area per unit length of stream.

which the criteria are presented, the type of information the criteria are based on, and how data are treated analytically.

Format

The format refers to the way the criteria are presented. The simplest format is binary (Fig. 4-17a), which brackets a range of a continuous variable (e.g., depth, velocity, distance from shore). Binary criteria act as simple on-off switches: the suitability index for a variable is 1.0 if it falls within the bracket and is 0.0 otherwise. Different ranges can be used to represent distinct categories of microhabitat quality for the target species. For example, a relatively narrow range could define habitat conditions preferred or selected by a life stage, or a more all-encompassing range could define conditions that the organism will use but not seek out. Binary criteria can even be used to describe conditions avoided by the target species. In this case, the output from PHABSIM would quantify unsuitable, rather than suitable, microhabitat.

In 1976, Waters proposed the use of the univariate curve (Fig. 4-17b) as a more robust alternative than binary criteria for expressing habitat suitability. Since then, the univariate curve has become the most familiar criteria format associated with PHABSIM. The tails of the curve are designed to encompass the entire suitable range of a continuous variable, but the narrow peak of the curve represents the optimum. Intuitively, the appeal of the univariate curve is that it is all-inclusive. Credit can be granted for conditions that are of intermediate habitat value, between optimal and barely useful. As popular as it is, however, the univariate curve has its detractors. Morhardt and Mesick (1988) summarized the complaints about univariate curves as follows:

1) When calculating the composite suitability index, univariate curves treat variables independently and potentially significant interactions between variables are ignored.
2) Weighted usable area, which results from the use of univariate curves, is an index and cannot be measured directly.
3) Different estimates of weighted usable area can be obtained by using different methods of aggregating the composite suitability index.
4) Weighted usable area combines elements of habitat quantity and habitat quality. A large area of low-quality habitat can produce the same weighted usable area as a small amount of high-quality habitat.

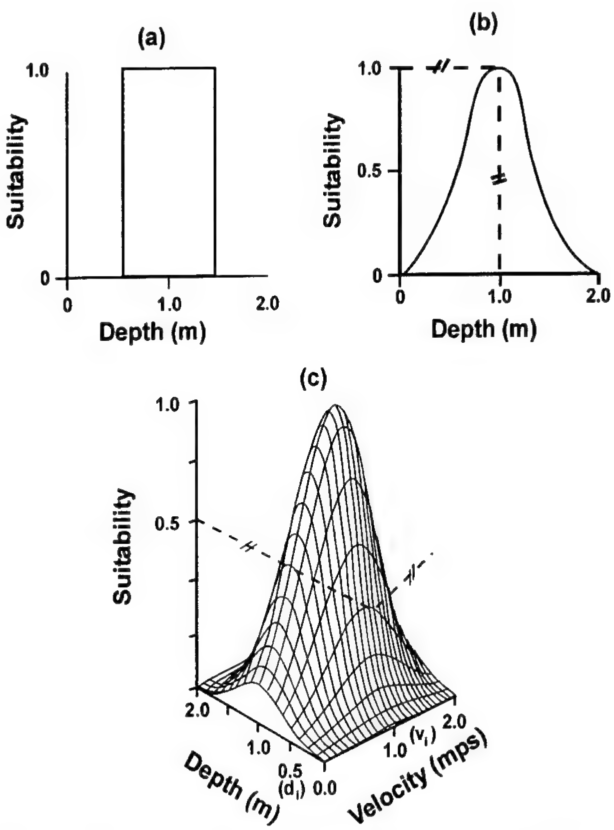

Fig. 4-17. Examples of three formats of habitat suitability criteria that can be used in PHABSIM: (a) binary, (b) univariate curve, and (c) multivariate response surface.

Multivariate criteria (Fig. 4-17c) overcome the problems of assumed independence and differential aggregation. A mathematical function is fit to frequency data for two or more variables at once and is often expressed in the form of an exponential polynomial equation:

$$P_{(d,v)} = \frac{1}{N} e^{-(a_1 d + a_2 v + a_3 d^2 + a_4 v^2 + a_5 dv)} \tag{13}$$

where $P_{(d,v)}$ is a joint probability of utilization for a combination of depth and velocity, N is a normalizing term reducing the area beneath the response surface to unity, and a_i are least squares parameters for the terms v, d, and dv.

The term $a_5 dv$ in equation 13 is called a cross-product and quantifies the correlation between the variables d and v in this bivariate model. As $a_5 dv$ increases in magnitude, the response surface will appear to twist in the x,y plane of Fig. 4-17c. The cross-product overcomes the assumption of independence because the variables in the model are treated jointly. The composite suitability index is computed directly by equation 13, so unlike univariate curves, there are no alternative aggregating functions. Because the composite suitability index can still have values between 0 and 1, however, the complaints about weighted usable area are still valid.

A disadvantage of the exponential polynomial is that it produces a symmetrical response surface with a single maximum value (Bovee 1986), which is unfortunate because habitat selection by fish often appears as a threshold function. For example, many fish species will use a wide range of depths without any apparent selective behavior once the depth exceeds some minimum value. This type of

behavior cannot be represented with an exponential polynomial and may be impossible for other multivariate functions as well. Because of these limitations and the fact that multivariate criteria are more difficult to use in PHABSIM than are univariate curves, criteria in this format have primarily been used as research tools rather than in routine applications of PHABSIM.

Some interactions among habitat variables that are known to be biologically important can be handled much easier using conditional criteria (Bovee 1986). An example of a biologically significant interaction is the use of deep water as a form of overhead cover by some species of fish. The fish will occupy shallow water areas if overhead cover is present, but they will only use deeper areas where cover is absent. This phenomenon can be depicted through the use of two depth suitability curves, one where overhead cover is present and one where it is absent. Relatively shallow water is assigned a suitability index of 1.0 for the with-cover situation. For the without-cover criteria, the depth curve does not reach a suitability of 1.0 until the water is much deeper. PHABSIM accounts for the two sets of criteria by making two simulation runs, one quantifying habitat for all the cells with cover and one for all the cells without cover. The results are then added together. Similarly, surface turbulence can be modeled as a form of overhead cover by using the Froude number (a ratio of potential to kinetic energy) instead of velocity to compute the composite suitability. In this case, cells without overhead cover would be unsuitable unless the Froude number was high enough to indicate surface turbulence.

Category

The category refers to the type of information and data treatment used to generate the criteria. It may come as a surprise to newcomers to PHABSIM, but it is just as valid to develop habitat suitability criteria for a mesohabitat type as for a species. For example, suppose that riffles are considered to be a critical mesohabitat type. A riffle could be defined with binary criteria, as having a depth range between 1 and 75 cm, velocities between 45 and 90 cm/sec, and a gravel/cobble substrate (or other definitions as negotiated among stakeholders). PHABSIM could then be used to determine how much area met those criteria over the range of simulated discharges.

Criteria that are derived from personal experience and professional opinion or from negotiated definitions are collectively known as category I criteria. These criteria can be developed relatively quickly and at minimal cost compared to more data-intensive approaches. Because the criteria are negotiable, obtaining consensus about the criteria may also forestall conflict over their subsequent use in PHABSIM. The principal drawback to category I criteria is that they are based on opinions instead of data. Credibility problems can arise when the criteria are presented to groups that were not involved in the original development process.

The least structured and most informal approach to developing category I criteria is the roundtable discussion. Scheele (1975) recommends three types of participants for roundtable discussions: stakeholders, experts, and a facilitator. It is important to include the stakeholders in this type of discussion, because they will be most immediately affected by the outcome of the study. In face-to-face discussions of this type, however, there can be a tendency for some stakeholder groups to "stack the deck" with their like-minded colleagues. Many can empathize with the lonely feeling of being the sole representative for a group, facing a phalanx of opposition from across the table. The group chairman or organizer should be careful to avoid overrepresentation by a group of stakeholders. The idea of a roundtable should be to encourage a diversity of experience, not to outvote the other side.

Experts who are familiar with the life histories and habitat requirements of the target species are also important participants in roundtable discussions. Often, some of the stakeholders can also be considered experts with regard to a particular species. However, recruiting a few more neutral experts from universities or research divisions of private or government agencies is a good idea. The first important quality about an expert is that he or she should be very knowledgeable about the habitat requirements of the target species. Second, experts must strive to be neutral and objective with respect to the criteria and its potential effects on the outcome of the study.

A facilitator may or may not be needed for a roundtable discussion, depending on the intensity of the institutional setting in which the study is being conducted. The role of the facilitator is to organize, clarify, and synthesize information, although the facilitator must also sometimes act like a referee. Even in amicable institutional settings, it may be advantageous to hire a professional facilitator simply for his or her organizational skills.

A more formal process of developing category I criteria is known as the Delphi technique (Zuboy 1981). Using this procedure, a small monitoring team devises a questionnaire, which is sent out to a larger respondent group of experts. After the questionnaire is returned to the monitor team, group opinion is summarized, usually by providing the median and interquartile ranges of the initial responses. The monitor team then provides the estimates of group opinion back to the respondents, who are asked to answer the questionnaire again in light of the new information. If a respondent's second response is outside the interquartile range from the previous round, he or she is asked to provide a brief explanation in support of the response. These explanations are then provided to the respondents in the next round, along with the revised median and interquartile ranges of the responses. The process is repeated until stability of the distribution of responses is achieved. Stability is not necessarily consensus but rather an indication that

the responses are not going to change very much, no matter how many additional rounds of response are conducted.

The Delphi technique lacks the rapid feedback and short response time of the roundtable method, but it also does not present problems of scheduling and repetitive meetings. Respondents can participate at their convenience, which may also mean that they will take more time to consider the advice they are giving. The anonymous nature of the Delphi technique also tends to counter the bandwagon effect associated with groups dominated by a few strong personalities. For this reason, minority opinions carry more weight in a Delphi exercise than in a roundtable discussion. The feedback loop in a Delphi exercise is long and slow, so it is important to be explicit and to avoid ambiguities. It is difficult to redirect a participant onto the right path once he or she ambles off on a tangent. Linstone and Turoff (1975) suggest that Delphi forecasts can be improved by using a blank questionnaire during the first round. For a criteria-development exercise, this would amount to providing the participants with copies of the suitability graphs with the axes labeled but without curves drawn on them. Participants would then be asked to sketch in the suitable and optimal ranges of the curves as their first response.

Category II criteria are based on frequency distributions of microhabitat attributes measured at locations used by the target species. These criteria are known as utilization or habitat use functions because they represent the conditions that were being occupied by the target species when the observations were made. This approach for criteria development dates back to the conceptual precursor of PHABSIM, a planimetric mapping method developed by the Washington State Department of Fisheries (Collings et al. 1972). The Washington method was designed to measure the amount of spawning area available to Pacific salmon at various streamflows. Because the methodology was oriented to spawning, criteria development consisted primarily of finding salmon redds and measuring depths and velocities at various locations around them. After a sufficient number of redds were measured, binary criteria were developed to encompass a specified range of the observations.

During the formative years of PHABSIM, the same basic approach for developing habitat suitability criteria was expanded to other species and life stages. One popular sampling method was to use a team of divers (Fig. 4-18) to intensively search many small reaches of stream to find locations occupied by the target species. At the end of each search, the depth, velocity, cover type, substrate, and other pertinent data were measured at each occupied location. After measurements had been taken on 100-200 locations, the investigator either defined a binary range for the criteria or fit the data to a univariate curve.

The benefit of category II criteria is that they are based on data, not on opinion. Error can be introduced to these criteria, however, through a bias of environmental availability. Manly et al. (1993) described this bias in the following way: even though a resource item is highly favored by a species, it will not be used much if the resource is hard to find. Conversely, less favored resource items will be used in larger proportion if they are the only ones available. In the context of microhabitat utilization, this bias means that individuals will be forced to use suboptimal conditions if optimal conditions are unavailable. By observing only the conditions used most often in a given stream, an investigator could confuse optimal microhabitat with conditions that were merely tolerable.

Category III criteria are designed to reduce the bias associated with environmental availability. These criteria are also referred to as electivity or preference functions. Resource selection refers to the use of resources disproportionate to their availability (Manly et al. 1993). For example, suppose that 10% of the stream mesohabitats occur as riffles, but 90% of the target species were found in riffles. This is an example of disproportionate use. In a slightly different definition, Johnson (1980) described preference as the likelihood that a resource will be selected if offered on an equal basis with others. The options for developing criteria vary somewhat depending on these definitions. For this reason, we have separated the discussions of electivity as defined by Manly et al. from those involving preference as described by Johnson.

A wide variety of mathematical indexes have been developed to indicate selection and, in some cases, avoidance of various resource units. The index of electivity usually involves a comparison of the proportion of the resource used with the proportion available or unused (available includes both used and unused proportions of the resource). The most familiar index of selection, at least with respect to habitat suitability criteria, has been the forage ratio:

$$E = \frac{U}{A} \tag{14}$$

where E is an index of electivity, U is the proportion of used habitat units of category i (for example, depths between 1.0 and 1.5 m), and A is the proportion of habitat units of category i available in the sample.

Other electivity indexes that have been used for category III criteria development include those developed by Ivlev (1961) and Jacobs (1974). Using the same terminology as equation 14, the Ivlev electivity index would be expressed as:

$$E = \frac{(U - A)}{(U + A)} \tag{15}$$

Fig. 4-18. A diver pulling himself along a rope, searching for rainbow trout in the South Platte River, Colorado. The upstream search allows the fish to be approached from behind, a technique designed to minimize disturbance and fright reactions.

and the Jacobs index as:

$$E = \frac{(U - A)}{(U + A) - 2UA} \qquad \textbf{(16)}$$

The purpose of these formulations was, in part, to bound the possible range of the electivity index between -1 and +1 and, in part, to distinguish selection from casual use or avoidance. Moyle and Baltz (1985) considered values between -0.25 and +0.25 using equation 16 to represent no preference. Values of E greater than +0.5 were interpreted to show strong preference, while values less than -0.5 indicated strong avoidance. In the context of binary criteria, these electivity indexes could be used to delineate microhabitat conditions defined as optimal (strong preference), usable (moderate preference), or merely suitable (anything not avoided).

Other methods that have been used for criteria development and testing include the chi-square goodness-of-fit test and principal components analysis. The use of these statistical tests will be illustrated in some of the examples that follow.

Sampling Designs

Manly et al. (1993) discuss several sampling protocols for determining resource selection. In one form or another, most of the approaches they describe have been used to develop habitat suitability criteria. The designs differ primarily in the manner in which resource use and availability data are collected. In the following section, we describe these sampling protocols and provide examples of how each was used for development of habitat suitability criteria.

Sampling Protocol A (SPA). Under this sampling design, all measurements are made at the population level. Used, unused, or available resource units are sampled or censused for the entire study area and for the collection of animals in the study area. Individual animals are not identified. For an illustration of SPA, refer to the work of Knight et al. (1991), who developed habitat guild criteria in the speciose streams of the Alabama River basin. They used a pre-positioned electrofishing grid (Fig. 4-19) to sample randomly selected locations throughout their study area. To take a sample, a grid was positioned at a predetermined, random location and left undisturbed for at least 15 min to allow fish to resume their normal activities. At the end of the waiting period, fish were collected by electrifying the frame for approximately 20-30 s. Immediately after activating the power supply, a seine was placed downstream of the frame so that stunned fish were swept into it. With the power active, one person kicked through the frame to dislodge benthic species. Fish captured at each location were

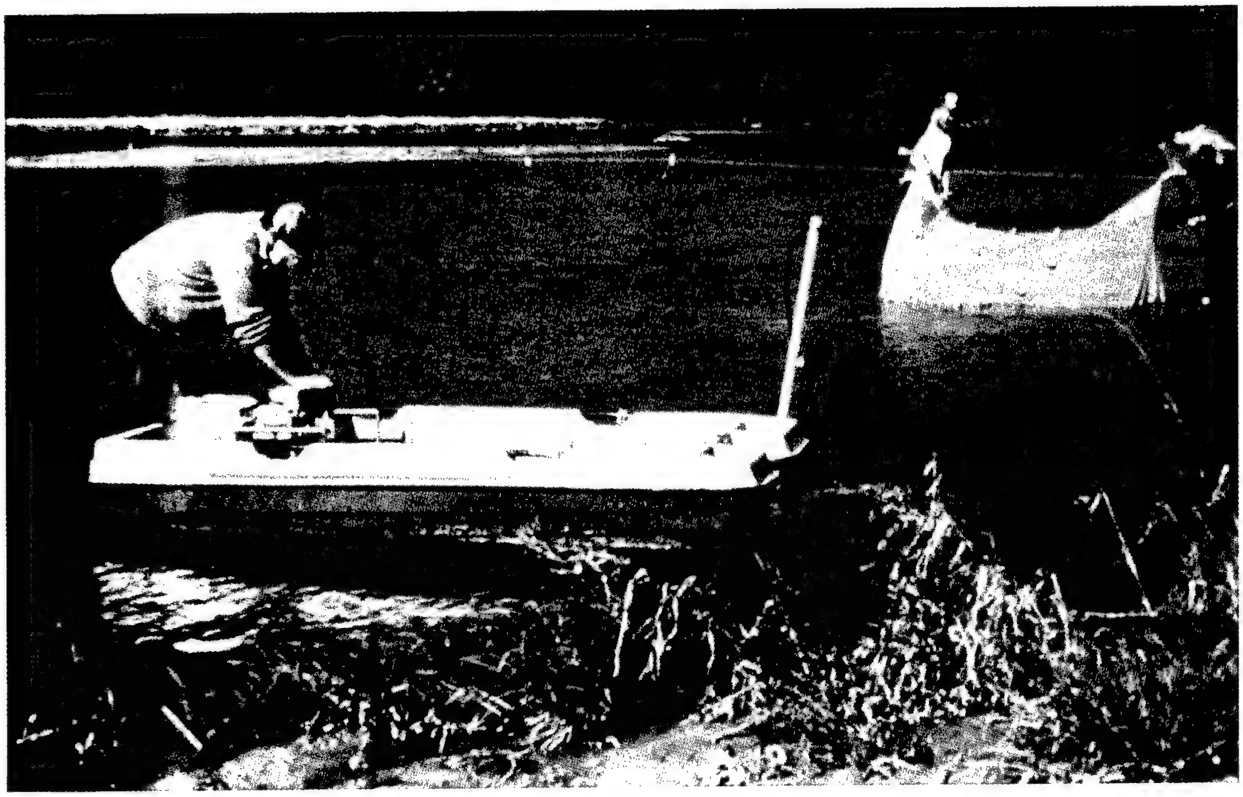

Fig. 4-19. Prepositioned electrofishing grid similar to the one used by Knight et al. (1991). Note the use of the beach seine to recover stunned fish downstream from the grid.

preserved for later identification in the laboratory. Habitat attributes (e.g., depth, substrate type, cover type, and velocity) were measured at the four corners of the frame for each sample and averaged.

A multivariate analysis of variance (MANOVA) was used to detect differences in habitat composition between samples that contained a particular species and those that did not. Principal components analysis (PCA) was used to illustrate the position of each fish species in habitat space (Fig. 4-20). These analyses were used to identify habitat attributes that consistently corresponded to high abundance and diversity of fish species. Variations of this method have also been described by Felley and Hill (1983), Bain et al. (1988), Scheidegger and Bain (1995), and Bowen (1996). The reader is directed to those articles for additional details regarding this method.

Sampling Protocol B (SPB). With this sampling design, individual animals are identified and the microhabitat locations they occupy are measured separately. Availability is measured at the population level. An example of this procedure is the use of a team of divers to make observations of the target species within the boundaries of a PHABSIM site (Fig. 4-21). This approach was used by Thomas and Bovee (1993) to derive preference functions for trout in the South Platte River, Colorado. Diving observations were conducted within the upper and lower boundaries of a PHABSIM study site. Teams of three to four divers conducted a complete census of the site, with the objective of identifying every location occupied by the target species. Weighted, numbered tags were dropped at each

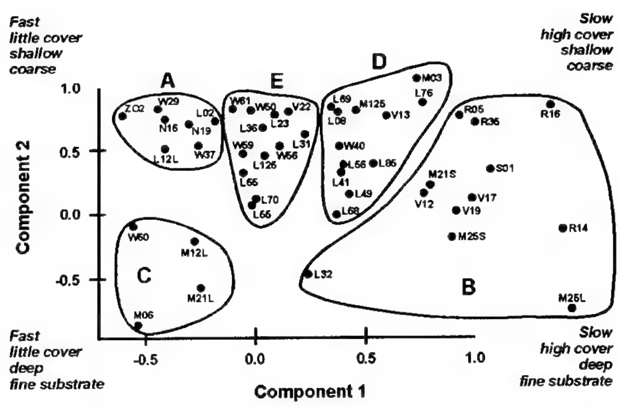

Fig. 4-20. Mean species locations in habitat space defined by principal components analysis of pooled data from seven streams in the Alabama River basin. Irregular circles encompass species with similar habitat use (e.g., Group C includes saddleback darter [*Percina vigil*], Alabama hog sucker [*Hypentelium etowanum*], black redhorse [*Moxostoma duquesnei*], and highfin carpsucker [*Carpiodes velifer*]. From Knight et al. (1991).

Fig. 4-21. A team of three divers conducting microhabitat observations in a PHABSIM site in the South Platte River, Colorado.

occupied location and data regarding the species, life stage, and activity (resting or feeding) were relayed to a data recorder on shore. At the completion of the dive, measurements of microhabitat variables (e.g., depth, mean column velocity, nose velocity, cover type) were made at each tag location and cross-referenced to the tag number.

As a completely separate activity, the hydraulic component of PHABSIM was used to estimate the distributions of depth, velocity, substrate type, and cover for the discharge present during the diving observations. This distribution was used to define the availability for each category of each microhabitat variable (e.g., 0.1 m depth increments). Use and availability data were pooled for all of the sites to obtain a single estimator of both. Electivity indexes were calculated using the forage ratio (equation 14) and normalizing the ratios so that the maximum value was 1.0 and the minimum was 0.0 (Fig. 4-22).

Sampling Protocol C (SPC). Habitat use and availability are sampled randomly throughout the study area with this sampling design. Each sample location is identified as either being used or unused by the target organism. Availability is defined as the combination of both used and unused locations. This approach is characterized by the use of a mobile anode electrofishing unit as used by Monahan (1991) to develop habitat suitability criteria for smallmouth bass (*Micropterus dolomieui*) and rock bass (*Ambloplites rupestris*) in the Huron River, Michigan. A crew of three to four people was needed for this sampling scheme. One person attended an electrofishing barge which contained

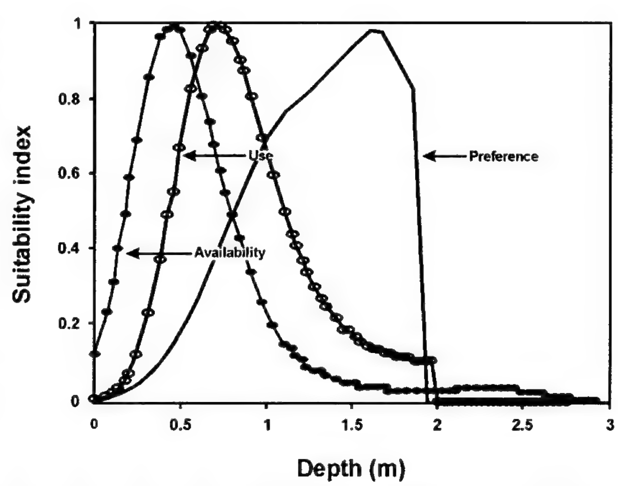

Fig. 4-22. Normalized depth use, availability, and preference functions for active adult brown trout in the South Platte River, Colorado. Preference was calculated using the forage ratio (equation 14).

a 3,500-watt generator, converted to 220-volt pulsed DC. The anode was attached to the variable voltage pulsator by a 15-m power cable. Another person carried the anode several meters upstream from the barge. Predetermined sampling locations were approached, with the barge located far downstream to minimize disturbance from the generator motor. When the shocking crew was in position, the person with the anode threw it in a high arch into the sampling location (Fig. 4-23). While the anode was in flight, it was energized by the attendant at the barge. As soon as the anode hit the water, one or two dip-netters rushed to the location and collected fish stunned in the field of the anode. At the conclusion of the collection, the netters noted whether the target species was present or absent in the sample, and microhabitat measurements similar to those described previously were taken. Microhabitat was measured regardless of whether the target species was present or absent. Habitat use was determined by aggregating all locations where the target species was present. Availability was determined by aggregating both occupied and unoccupied locations, and electivity was calculated using equation 14.

Sampling Protocol D (SPD). This approach is a variation of SPB and SPC. Habitat use is determined either by complete census (e.g., SPB) or sampling (e.g., SPC), but rather than using availability, a random sample of unused habitats is taken. This approach is exemplified by the method devised by Thomas and Bovee (1993) to test the transferability of habitat suitability criteria from one stream (source) to another (destination). Habitat use was determined by diver observations, following the same procedures described for SPB. In addition, 25 unoccupied locations in each dive site were selected at random and marked with a coded tag. Microhabitat measurements and occupancy data were recorded for each tag.

Criteria to be tested were converted from univariate curve to binary format, with the optimum range for a variable defined as having a composite suitability index greater than 0.85. Usable microhabitat was defined as having a composite suitability value between 0.2 and 0.85. Suitable microhabitat was defined as the full range of conditions in which the target organism was observed. Unsuitable microhabitat was defined as all values outside the suitable range. From the criteria, each used and unused location was classified as being optimal or usable and suitable or unsuitable.

Then, each sampled location was cross-classified (e.g., occupied-optimal, unoccupied-usable) into a 2 x 2 contingency table (Table 4-4). The null hypotheses H_{01} (optimal locations will be occupied in the same proportion as

Fig. 4-23. Mobile anode electrofishing system used in the Huron River, Michigan, to develop habitat suitability criteria for smallmouth bass and rock bass (Monahan 1991).

Table 4-4. Contingency table format for one-sided chi-square test of optimal versus usable classifications of microhabitat.

	Optimal	Usable	Total
Occupied	a	b	a + b
Unoccupied	c	d	c + d
Total	a + c	b + d	N

usable locations) and H_{02} (suitable locations will be occupied in the same proportion as unsuitable locations) were tested using a one-sided chi-square test (Conover 1980). The test statistic T was given as:

$$T = \frac{\sqrt{N}\,(ad - bc)}{\sqrt{(a+b)(c+d)(a+c)(b+d)}} \tag{17}$$

where N is the total number of measured locations, a is the number of occupied optimal locations, b is the number of occupied usable locations, c is the number of unoccupied optimal locations, and d is the number of unoccupied usable locations. Suitable locations were substituted for optimal locations and unsuitable locations for usable to test classifications of suitable and unsuitable microhabitat.

For a set of criteria to be considered transferable, both null hypotheses should be rejected at the 0.05 level of significance. (Note: The critical value of T at this significance level is 1.6449 and is obtained from the normal distribution table, *not* from the chi-square distribution. See Conover [1980] for discussion.)

Sampling Protocol E (SPE). This approach originates from the definition of preference given by Johnson (1980), who described preference as the likelihood that a resource will be selected if offered on an equal basis with others. Conceptually, this definition would apply to habitat suitability criteria if it were possible to develop criteria in a stream with a perfectly uniform distribution of microhabitat conditions. According to Johnson's definition, one could determine habitat preference simply by measuring use, because all microhabitat conditions would be equally available. Obviously, streams fitting this description are few and far between (if not nonexistent). However, it may be possible to approximate this condition by altering the sampling design when collecting habitat use data. Following this strategy, each mesohabitat type would be sampled equally, regardless of its relative abundance in the stream. The premise underlying the concept of equal-effort sampling is to equalize the availability of microhabitat conditions within the sampled subpopulation, to the extent possible. Thomas and Bovee (1993) collected equal-effort habitat use data for comparison with data collected by proportional sampling. During this part of the study, five mesohabitat types were identified: riffle, run, shallow pool, deep pool, and pocket water. Sites were selected to minimize the amount of redundancy in microhabitat conditions, and the lengths of the sites were adjusted to equalize their surface areas. Fish observation data were collected by surface diving, using the same technique illustrated in Fig. 4-21, but no availability or unused habitat measurements were taken. The data were then fit to univariate curves, following the procedures described for category II criteria.

For this data set, the greatest discrepancy among curve categories occurred among the respective depth curves. For example, the depth curve for active adult brown trout derived using equal-effort data shifted slightly to the right when contrasted with the category II curve obtained from proportional sampling (Fig. 4-24). In comparison, however, the curve derived from equal-effort sampling did not shift to the right as much as the category III curve derived from a forage ratio calculation. We believe that the "adjustment" from equal-effort sampling will usually be less extreme than using an electivity index, but more research is needed to support this assertion.

Manly et al. (1993) also debated the advantages of observing a single episode of selection versus pooling observations from several periods of selection. The only obvious conclusion was that the analysis is a lot easier if only a single episode is observed. Beyond that, the answers become fuzzier. From the perspective of developing habitat suitability criteria, observing a single selection episode amounts to collecting all habitat use data at the same discharge, during a time interval small enough to ensure

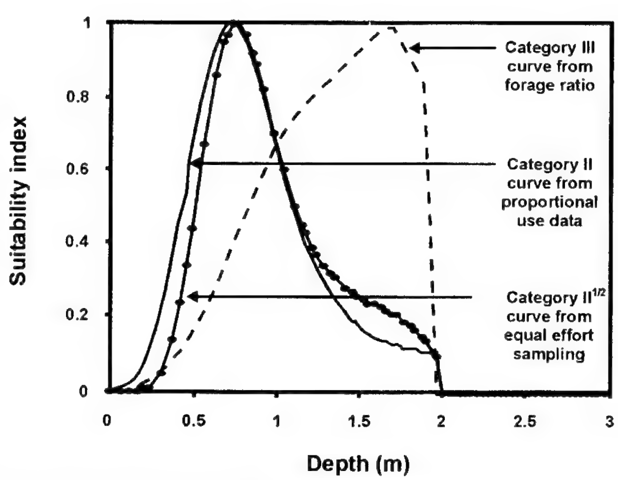

Fig. 4-24. Comparison of depth suitability curves for active adult brown trout derived from proportional use data (category II), equal-effort sampling (category II½), and forage ratio correction for availability (category III).

consistent behavior of the target organism. Although it is easier to assemble criteria from a single observation episode, doing so may not be possible, nor will it necessarily result in the most accurate criteria. For example, obtaining habitat use data from radio telemetry is sometimes necessary. This technique is particularly useful for studying species that are rare, cryptic, or otherwise hard to sample or observe. It is impossible to monitor more than a few frequencies at once (generally 10-25), and battery life for transmitters implanted in fish is usually fairly short. These circumstances almost guarantee that the habitat use of the same animal will be measured more than once. Otherwise, the sample size of observations will be too small for criteria development. If the time between observations of the same animal is too small, the assumption of independence may be violated, resulting in pseudoreplication (Hurlbert 1984). If the time period is too long, however, either the battery will die or the behavior of the animal will likely change.

Whether to try to make all observations at one discharge or to spread them out over several streamflows is a serious consideration. By measuring habitat use over a range of flows, the universe of sampled conditions will be expanded. For category II criteria, collecting data over a range of flows seems to have merit. For equal-effort sampling, measuring

Observations About Habitat Suitability Criteria

- In PHABSIM, it is as important to describe habitat variables used over a broad range as it is to find narrow habitat preferences. Sometimes, these criteria appear as thresholds, above or below which there is little selection. The empirical frequency distribution of utilized microhabitat may be artificially narrow, either because the range of available conditions was narrow or because of the method used to fit the criteria to the data. Such artificially narrow criteria can make PHABSIM output unrealistically sensitive to changes in discharge. If this phenomenon is suspected, it should be brought to the attention of the stakeholder group. Modifying a data-based curve on the basis of professional judgement is acceptable, if not encouraged.
- Category I criteria are as valid in an application of IFIM as data-based criteria, if they are supported by a consensus of opinion among the stakeholders. Criteria developed by a committee are often somewhat broader than those based on data, because the ranges of the criteria are often negotiated. In this respect, category I criteria rarely exhibit artificial narrowing but can be too all-inclusive. If the criteria are too broad, PHABSIM output may be relatively insensitive to changes in discharge.
- Sampling protocol A differs most dramatically from the other methods we discussed. A unique advantage of SPA is that it can be used to define habitat use guilds and critical habitat types. This advantage allows the analyst to concentrate on only a few key habitat types, rather than analyzing habitat for multiple species and life stages. Furthermore, potential biological connections between the habitat type and the biology of the stream can be quantified through the statistical analysis.
- It is possible to develop category I criteria for critical habitat types similar to the kind of criteria one would obtain using SPA. Rather than asking experts to define suitability criteria for different life stages of a species, they can be asked to identify habitat types that they consider to be important. Not only can the experts identify such habitat types, but they can also tell you why the types are important. If the experts cannot put suitability index numbers on a set of criteria to represent the habitat types, go out to a river, have the experts identify the critical habitat types, and measure the habitat attributes directly. (This method is known as BOBSAR—a Bunch of Biologists Standing Around a River.)
- There is no unbiased way of collecting habitat use data. Every kind of sampling gear or observation technique has some sort of limitation that can result in biased data. For example, even with good visibility, divers are more apt to spot fish in shallow water than in very deep pools. They are also more likely to find active fish than those at rest, because the human eye is attracted to movement. As another example, electrofishing is less effective in deep water and for safety reasons may be restricted to depths less than 1 m. When evaluating criteria, consider the limitations of the methods used to obtain the data in the context of the source stream. If you believe that the criteria are unduly biased, refer to the first observation.
- One of the assumptions related to the estimation of resource selection functions is that the variables which actually influence selection have been correctly identified and measured. Perhaps the most common misconception about PHABSIM is that the only microhabitat variables that can be used are depth, mean column velocity, and substrate. In fact, PHABSIM can accept a wide range of variables, provided that they are related to the hydraulics or structural characteristics of the stream. Examples of alternative microhabitat variables include nose velocity, adjacent velocity (important for feeding stations), cover type, distance to cover, distance from shore, proximity to another habitat type, sheer stress, Froude number, depth-velocity products (important for recreational activities),

different mesohabitats at different flows might actually help eliminate microhabitat redundancy among sites. One of the assumptions associated with derivation of resource selection functions, however, is that organisms have free and equal access to all available resource units (Manly et al. 1993). For category III criteria, this assumption might be violated if data from low flows are pooled with those from high flows. The same conditions are probably not available during both periods. To overcome this problem, Locke (1988) suggested developing separate category III curves for high and low flows and averaging them.

Channel Structure and Hydraulics Components

Data Collection

The first step in collecting physical data for PHABSIM involves site layout and preparation. During site layout, transects and stream cells are placed at strategic locations to define microhabitat characteristics of the site. The specificity or detail with which the site is defined depends on the number of transects used and how finely they are dissected across the stream. Proper site layout, like beauty, often exists in the eye of the beholder (in this case, those laying out the site). In a broad sense, it may be better to err

and depth as a form of cover. Additionally, PHABSIM can be used in conjunction with commercially available spreadsheets to derive alternative habitat indexes, such as habitat diversity. Be aware that you are not limited to the "default" variables. With a little imagination PHABSIM can usually be tweaked to include microhabitat attributes known to be important to the target species.

- Criteria verification is always a good idea. Remember that in IFIM, the currency for negotiations is total habitat and alternatives are evaluated by comparison with a baseline. Thus, habitat does not necessarily have to have a demonstrated biological connection, as long as the currency is meaningful to the stakeholders and decision-makers. For negotiations of low to medium levels of conflict, criteria can be verified simply by obtaining their acceptance among the stakeholders. When it becomes impossible to obtain consensus among the stakeholders, however, it may be necessary to conduct an empirical verification test. If PHABSIM sites are already established, a relatively easy way to verify the criteria is to measure the standing crop in each site, calculate the habitat availability at the same discharge, and relate the two statistically (e.g., Fig. 4-25). This procedure has two advantages. First, a significant correlation is unlikely if the criteria are incorrect, so the chances of committing a Type I error (accepting criteria when they should be rejected) are fairly remote. Second, a significant correlation lends credence to the idea that microhabitat is biologically meaningful. The disadvantage of this procedure is that the correlation may not be significant for reasons other than inaccurate criteria. One often overlooked reason for poor correlations is the error associated with the estimate of standing crop. Before getting too excited about a lack of significant correlation, it is a good idea to examine the confidence intervals for the population estimates at each site. You may find that differences in standing crop cannot be quantified reliably enough to perform this type of test. A related problem occurs when trying to validate criteria for sport fish, because places that have good fish habitat also attract fishermen. Sometimes, sites with the best and most abundant microhabitat will have the smallest standing crop, simply because all the good habitat may be "fished out."

  The approach outlined by Thomas and Bovee (1993) can be performed without having to establish a PHABSIM site and it is less influenced by fishing pressure than the previous method. Also, the results of the Thomas-Bovee method are not dependent on accurate estimates of standing crop. The primary disadvantages of this method are that it requires relatively unbiased habitat use and nonuse data, and the results are meaningful at the individual level but not necessarily at the population level. In other words, the criteria may be transferable, in the sense that they describe fish behavior, but there is no guarantee that habitat estimates from the criteria are related to fish numbers.

- So how should you proceed with the acquisition of accurate and realistic habitat suitability criteria for your application of IFIM? As with everything else associated with this methodology, there is no one best way that fits every circumstance and situation. You should discuss all the options among your fellow stakeholders and with other professionals who have had experience with IFIM. Consider whether there are sanctioned sets of criteria or standard operating procedures in the state you are working in. Whatever you do, decide on a course of action and then stick to it. What you should not do is proceed with the study using several sets of criteria, deferring the decision about which is correct until you see the results from PHABSIM. Our experience with this approach is that it merely mires all of the stakeholders in their previously held positions.

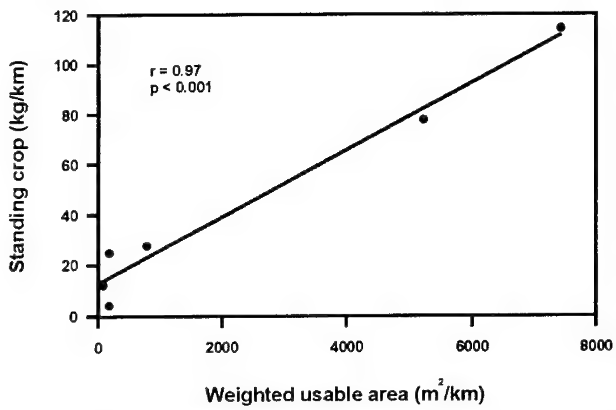

Fig. 4-25. Relation between weighted usable area and standing crop of cutthroat trout (*Oncorhynchus clarki*) for six sites in Yellowstone National Park.

on the side of too many transects than not enough. Study implementors, however, must be careful not to overcompensate during site layout. Describing sites too intensively may restrict the ability to measure replicate sites.

An effective technique for site layout is to use a stratified random sampling approach for transect placement. The stratifications are defined by establishing longitudinal cell boundaries, short lengths of stream in which the slope, bed topography, substratum, hydraulic characteristics, and cover distribution are all relatively homogeneous (Fig. 4-26). Placement of cell boundaries can be fairly subjective, so this is a good time for a riverside committee meeting among stakeholders and field crews. Once the cells are established, however, transect placement is easy. Purists in sampling theory will place transects randomly within cell boundaries. Nonpurists skip this step and simply place the transect in the center of the cell. Theoretically, if the cell is really homogeneous, it should not matter what method is used for transect placement.

Another significant part of establishing a PHABSIM site is identifying and measuring all of the hydraulic controls in the site. Recall that a hydraulic control is a constriction in the channel that creates a backwater effect in an upstream direction. The crest of a riffle is the most common type of vertical constriction occurring in natural channels. Features that cause abrupt narrowing of the channel, such as bedrock outcrops, may form horizontal constrictions. Vertical constrictions are more effective as hydraulic controls at low flows, but horizontal constrictions may become more effective at high flows. Hydraulic controls are important because of the input requirements of the hydraulic simulation models in PHABSIM. The IFG4 model, for example, relates water surface elevations and discharges in order to predict depths across the channel at unmeasured flows. The lowest point on a riffle defines the water surface elevation at upstream transects when the discharge is zero (Fig. 4-27). This elevation is known as the stage of zero flow (SZF). In IFG4, the SZF defines the y-intercept of the relationship between water surface elevation and discharge.

The Water Surface Profile (WSP) program calculates the water surface elevation at a transect on the basis of the water surface elevation of the adjacent transect downstream. In order to initiate a simulation, it is necessary to provide an estimated water surface elevation at the downstream-most transect for any simulated discharge. Most often, these initial water surface elevations must be predicted, and the only places in the stream where they can be predicted reliably is at a hydraulic control.

Once the stream cells and transects have been installed, a horizontal and vertical reference system must be established to prepare the site for cross-sectional and hydraulic data collection. The horizontal reference system determines the positions of transects and stream cells relative to one another. Establishing horizontal control can be as simple as measuring distances between transects and cell boundaries or as complicated as drawing a scale planimetric map of the entire study site.

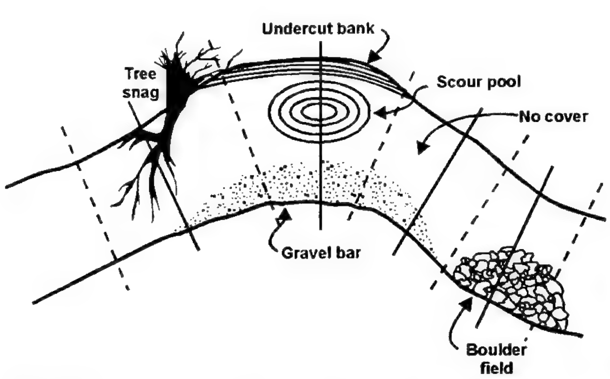

Fig. 4-26. Establishment of longitudinal cell boundaries according to the distribution of various cover types, substratum, and topographic features. Notice that "No Cover" is considered a cover type.

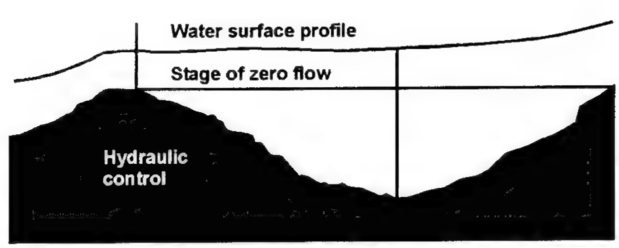

Fig. 4-27. Longitudinal profiles of stream thalweg and water surface, illustrating how a vertical constriction (riffle) behaves as a hydraulic control. The lowest elevation across the riffle (i.e., the thalweg evaluation) is the stage of zero flow.

Vertical control is instituted by determining the elevations (usually relative to an arbitrary reference elevation or datum) of numerous benchmarks in the site. The benchmarks are used in conjunction with differential leveling, by which all of the ground and water surface elevations in the site can be tied to the common reference elevation. Precise vertical control is an important aspect of quality assurance in PHABSIM data collection. If the elevations in a site are not all tied to the same datum (or if there was considerable error in establishing vertical control), hydraulic simulations may be adversely affected. Poor vertical control results in increased simulation error and limited simulation range.

The next step in the data collection sequence is usually to measure cross-sectional profiles. Many practitioners collect profile data prior to or coincidentally with hydraulic data, but profile data really can be collected at almost any stage of the process. In PHABSIM, a channel cross-section is described as a series of verticals, comparable to those used in discharge measurements. Each vertical is described by (1) a distance from a known point across the channel, (2) the ground elevation at that distance, and (3) descriptors of structural cover and substrate associated with the location.

For streams of small to moderate size, cross-channel distances are typically measured using a measuring tape or tagline (a rope or cable marked at increments), and ground elevations are determined by differential leveling. In large rivers, where part or all of the measurements are made from a boat, distance measurements are taken from a static line (essentially a very heavy duty tagline) to which the boat is attached. Ground elevations in the active channel are determined by sounding, whereby the elevation of the streambed is found by subtracting the depth from the water surface elevation (Fig. 4-28). Sounding is somewhat less accurate, but considerably safer and more practical, than trying to manipulate a leveling rod in deep, swift water. Differential leveling is used to determine the water surface elevation and all of the ground elevations above the water level.

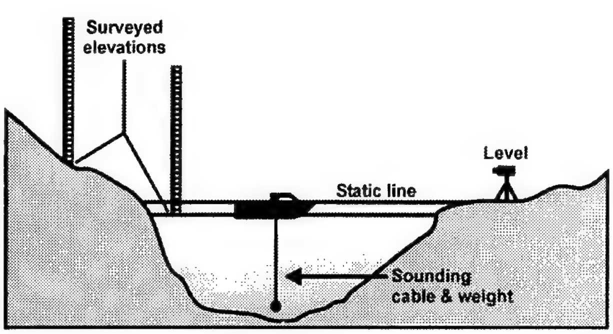

Fig. 4-28. Measurement of a cross-sectional profile using a combination of differential leveling and sounding techniques.

In addition to cross-sectional distances and elevations, channel profile data include descriptors of substrate and cover at each vertical across the cross-section (including verticals that lie above the water surface). The level of detail necessary for describing channel characteristics is dictated by the detail of the habitat suitability criteria for the target species. If the criteria have not been verified prior to PHABSIM data collection, it is better to describe the channel characteristics with too much detail rather than with too little. Complex descriptions can be simplified, but simple descriptions cannot be made more detailed.

Describing channel characteristics for PHABSIM can be impaired by restricted visibility, especially in streams that are too deep to wade. In small streams, it may help to collect channel data at a relatively low flow because more streambed will be exposed and the water may be more transparent. In large, turbid rivers, substrate typing can be performed by using a scientific echo sounder, which analyzes the characteristics of the echo signature to distinguish hardness and irregularity of the streambed. About the only restriction associated with echo sounders is that bottom-typing is marginal in depths less than 1 m or so. The instruments are not affected by turbidity, but the resolution of substrate types is somewhat restricted. Our experiences with the echo sounder indicate that mud, sand, gravel, and bedrock can be identified quite well, but distinguishing between gravel, cobble, and small boulders is difficult. We have not found this type of echo sounder to be very useful for identifying submerged cover objects. Side scanning sonar would probably be better for this purpose.

The final step in collecting PHABSIM data is to measure the calibration velocities and water surface elevations. We advise that hydraulic calibration data should include at least three water surface elevation-discharge data pairs and one set of calibration velocities. High and low calibration discharges should differ by at least a half (and preferably, a full) order of magnitude. In complex channels, such as those containing islands and multiple side channels, it is advisable to measure water surface elevations at five or six discharges. Several calibration pairs should be taken at low flows, when standing backwaters occur in the side channels. Additional calibration pairs are recommended over a range of discharges when the side channels are flowing. In order to simulate side channel habitats accurately, it is also prudent to measure the distribution of the total discharge among the side channels.

PHABSIM relies on an empiricism for its velocity predictions. Therefore, it follows that the accuracy of the velocity predictions will improve with increasing amounts of calibration data. Calibration velocities, however, are some of the most expensive data to acquire, so it is common practice to measure velocities at only one or two discharges. Because of the way PHABSIM simulates velocities, downward extrapolations (e.g., simulating discharges lower than

the calibration discharge) are more accurate than upward extrapolations. When extrapolating upward, some of the verticals may have been above water at the calibration flow and were uncalibrated for the simulated discharge (Fig. 4-29). Therefore, if only one set of calibration velocities are to be collected, it is generally better to measure them in the mid-flow range (e.g., ½ to 2/3 of bankfull) to avoid extrapolating upward over a large range of discharges. In channels exhibiting complex velocity distribution patterns, two sets of calibration velocities will help improve accuracy of velocity predictions. Generally speaking, the velocity distribution will typically achieve its greatest complexity at low flows, so it follows that if a second set of calibration velocities is to be measured, the data should be collected at a relatively low discharge. The low-flow calibration velocities would be used primarily to simulate lower discharges, whereas the moderate-flow calibration would be used for simulating higher flows.

Calibration

Not counting the verification of habitat suitability criteria, calibration in a PHABSIM analysis is primarily confined to the hydraulic simulation component. There are two separate but related calibration/simulation activities: (1) the calibration and simulation of water surface elevations and (2) simulation of mean column velocities.

Water surface elevations. Water surface elevations can be generated by four component hydraulic models: IFG4, MANSQ, WSP, or the U.S. Army Corps of Engineers' HEC-2. It is also possible to mix these models to take advantage of their individual strengths (Fig. 4-30).

IFG4 uses an empirically derived rating curve, very similar to that described for a gaging station, as its primary water surface elevation predictor. A least-squares regression is fit to three or more pairs of log-transformed, water surface elevation-discharge data. The water surface elevation for an unmeasured (simulated) discharge is found by interpolation or extrapolation along the regression line (Fig. 4-31). The actual regression is performed on the water surface elevation minus the stage of zero flow. This model ensures that there will be standing water in pools when the discharge is zero.

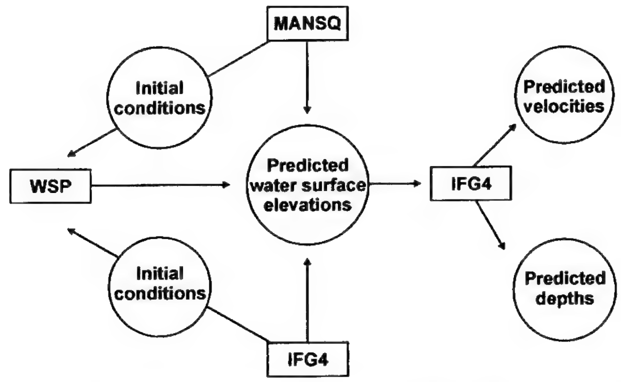

Fig. 4-30. Various configurations by which hydraulic simulation programs are mixed and matched, and information flow from one program to another in the hydraulics component of PHABSIM.

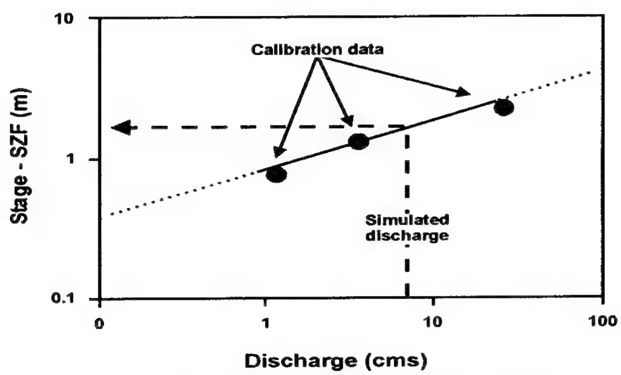

Fig. 4-31. Log-transformed rating curve used to predict water surface elevations at unmeasured discharges in IFG4. The algorithm in IFG4 relates the discharge to the water surface elevation (stage) minus the stage of zero flow (SZF). This formulation ensures that pools will have standing water at zero discharge.

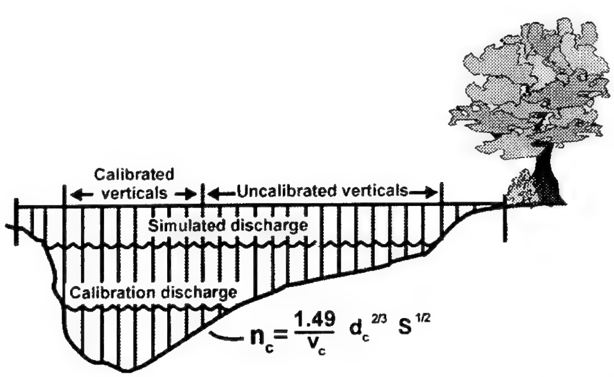

Fig. 4-29. Calibration and simulation of the velocity prediction algorithm in the IFG4 hydraulic simulation program.

IFG4's rating curve approach is not restricted to any particular stream setting. This characteristic and the inherent simplicity of the approach are probably the greatest strengths of the model. IFG4's weakness is the assumed straight-line relationship between the log-transformed water surface elevations and discharges. Streams rarely exhibit such linear relations throughout their entire range of flows. More commonly, the overall rating curve is usually curvilinear and only portions of it can be approximated by linear segments. In some streams, the linear segments are so short that IFG4 becomes an impractical, if not invalid, model. Consequently, even though we can say that there are no limitations on where IFG4 will work, there are limitations on where it will work well.

As its name implies, the MANSQ program uses the Manning equation to determine water surface elevations at simulation discharges. In this case, the version of Manning's equation used is:

$$Q = C\,R^{2/3}\,A \qquad (18)$$

$$C = \frac{1.49}{n}\sqrt{S} \qquad (19)$$

where Q is the discharge, R is the hydraulic radius (cross-sectional area divided by wetted perimeter), A is cross-sectional area, n is Manning's roughness coefficient, and S is the energy slope at the section.

Input to the program includes the cross-sectional profile, a calibration discharge (CALQ), and a corresponding water surface elevation. Given the water surface and cross-sectional profile, MANSQ calculates the hydraulic radius (R) and the cross-sectional area (A) and determines the conveyance factor (C). Because Manning's n (and to a lesser extent, energy slope) varies as a function of discharge (Fig. 4-32), however, C varies exponentially as:

$$C_q = (C_{cal})^{\beta} \qquad (20)$$

where C_q is the conveyance factor at a simulated discharge, C_{cal} is the conveyance factor at the calibration discharge, and β is the slope of a regression line relating C to discharge. During calibration, the user inputs any additional flows for which calibration water surface elevations exist. β is then adjusted until the water surface elevations predicted by MANSQ match the measured water surface elevations reasonably well.

MANSQ employs an iterative solution technique to find the water surface elevation that corresponds to a simulated discharge. MANSQ contains an algorithm that provides an initial estimate of this elevation. A value for β corresponding to the simulated discharge is interpolated or extrapolated, and equation 20 is solved to find C_q. C_q is substituted into equation 18 to determine Q. If the discharge estimated from equation 18 (Q_{est}) is larger or smaller than the discharge initially entered as the simulation flow (Q_{sim}), the water surface is lowered or raised, respectively, and the calculation sequence repeated until convergence is achieved ($Q_{est} = Q_{sim}$).

In some channels, particularly those with a triangular cross-section, the cross-sectional area and discharge can vary nonlinearly with incremental changes in stage. These nonlinear relationships can cause a great deal of trouble in IFG4, but they are accounted for in MANSQ. MANSQ's most important limitation is that it can only be used at channel cross-sections that are not influenced by backwater effects. In practical terms, this limitation constrains the use of MANSQ primarily to riffles and other mesohabitats that do not exhibit backwaters. MANSQ is ideally suited for determining water surface elevations on hydraulic controls, however, so one of its primary uses is to provide initial conditions to the WSP or HEC-2 model.

WSP and HEC-2 share many of the features of the MANSQ program, with one important exception. Whereas MANSQ cannot be used where backwaters exist, both WSP and HEC-2 were designed specifically for backwater applications. These programs are generically termed "step-backwater" models and employ a technique known as energy balancing to predict water surface elevations in a backwater.

In a step-backwater model, the total energy at a transect for a given discharge is the sum of the potential energy and kinetic energy at that point along the stream (Fig. 4-33). The potential energy is determined by the elevation of the water surface above sea level (or an arbitrary datum). In Fig. 4-33, this elevation is denoted as WS1 at transect 1 and WS2 at transect 2. The kinetic energy is represented by the expression:

$$\frac{V^2}{2g} \qquad (21)$$

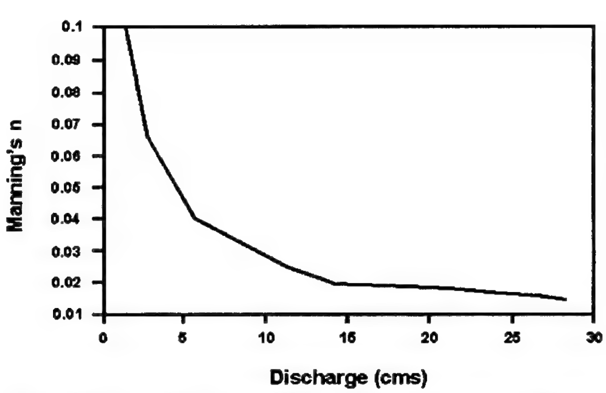

Fig. 4-32. Example of variable Manning's n phenomenon. Because n does not change much after the depth becomes large relative to the streambed materials, the most dramatic changes in n-values occur in the low-flow range.

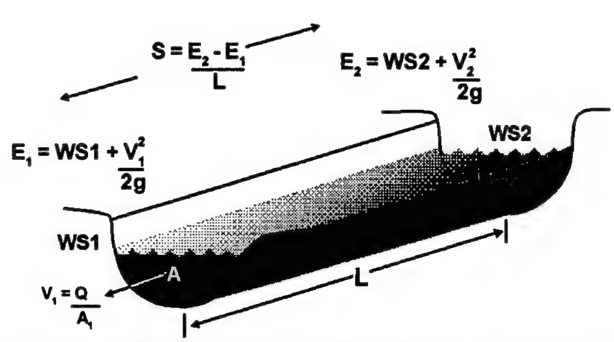

Fig. 4-33. Determination of energy slope (S_e) between two transects, using the Bernoulli and continuity equations. L is the distance between transects, E_i is total energy at a transect, V_i is the average velocity for a transect, WS_i is the water surface elevation at a transect, A_i is the cross-sectional area at a transect, g is the acceleration of gravity, and Q is the discharge.

where V is the average velocity for the cross-section (determined by dividing the discharge by the cross-sectional area) and g is the acceleration of gravity. The calculation of total energy, as shown in Fig. 4-33, is known as the Bernoulli equation:

$$E = (B + D) + \frac{V^2}{2g} \tag{22}$$

where E is the total energy at the transect, B is the elevation of the streambed, and D is the depth (note that the bed elevation plus the depth equals the water surface elevation).

Equation 21 expresses kinetic energy in the same linear units as potential energy (square meters per square seconds divided by meters per square second is meters). Hydraulic engineers refer to this expression of total energy as head. Head loss refers to the amount of total energy dissipated as a mass of water moves from one transect to another, and the energy slope is calculated as the head loss divided by the distance between the two transects.

The energy gradient can also be calculated using Manning's equation. By rearranging equations 18 and 19, and solving to find S, we obtain:

$$S = \frac{Q^2 n^2}{(1.49)^2 R^{4/3} A^2} \tag{23}$$

In spite of the impressive series of equations involved, the way a step-backwater model works is not very different from the iterative solution of MANSQ. Input is provided in the form of a discharge, an initial water surface elevation at the first transect, and an estimate of Manning's n for transects 1 and 2. WSP (or HEC-2) then estimates a water surface elevation at transect 2. From the estimated water surface, the model calculates the hydraulic radius, cross-sectional area, and mean channel velocity for each cross-section. These parameters are then used to calculate the energy gradient between the two sections using both the Bernoulli and Manning equations. If the energy gradient is not the same when calculated by both equations, the water surface elevation is adjusted at transect 2 and energy losses recalculated. This process is repeated until the same energy loss is calculated using both equations. The solution obtained in this fashion is said to be energy balanced. Having determined the water surface elevation at transect 2, WSP uses the elevation there as a new starting point and determines the water surface elevation at the next upstream transect. In this fashion, the "backwater is stepped upstream," hence the generic name of the model.

Up until this point, the investigator is usually blithely unaware of the thousands of computations being performed in his or her computer. Active manual calibration starts when the energy-balanced water surface elevations are compared against measured elevations. If predicted and measured water surface elevations do not match reasonably, Manning's n is changed to raise or lower the predicted water surface elevations. The goal of this activity is to obtain a reasonable fit between predicted and measured water surface elevations. What constitutes a reasonable fit is often determined by the characteristics of the stream being simulated and the inclination of the investigator. Obtaining a reasonable fit is step 1.

Step 2 involves recalibration at alternate discharges to account for the variable roughness phenomenon (Fig. 4-32). Step 2 is essentially the same as step 1, except that a new discharge and starting water surface elevation are provided to the model and global modifications are made to Manning's n by multiplying the n-values from step 1 by a roughness modifier. During step 2, these roughness modifiers are adjusted until reasonable agreement between observed and predicted water surface elevations is again achieved. Step 2 is repeated for however many additional sets of calibration data are available. The final activity under step 2 is to develop a relationship between simulated discharge and roughness modifier (e.g., by linear regression of logarithmically transformed values).

WSP and HEC-2 require much more hands-on calibration and tinkering than empirical models like IFG4. Step-backwater models also perform poorly in streams that have abrupt changes in slope over relatively short distances (e.g., cascade-plunge pool sequences). Successful simulations can be obtained, however, by dividing the input data into discrete combinations of hydraulic control and pool transects. The advantage of step-backwater models is that they generally provide better predictions of water surface elevations over a wider range of conditions and discharges than either IFG4 or MANSQ. Step-backwater models are superior for simulations of out-of-bank flow conditions. If it is necessary to simulate flood flows, WSP or HEC-2 will probably be necessary.

Velocities. Velocity predictions in PHABSIM are usually performed by IFG4. The measured velocity at a vertical is used to calibrate a modified version of the Manning equation:

$$n_i = \frac{1.49}{V_i} d_i^{2/3} S^{1/2} \tag{24}$$

where n_i is a roughness coefficient (Manning's n) for vertical i, V_i is the calibration velocity for vertical i, d_i is the depth of the vertical (found by the difference between streambed and water surface elevations), and S is the energy slope (approximated by the hydraulic slope) at the cross-section.

When another discharge is simulated, IFG4 obtains a water surface elevation corresponding to the new discharge, either by way of its internal rating curve or from MANSQ, WSP, or HEC-2. The new water surface elevation results in new depths (d_i) for all of the verticals. The new depths are then substituted back into Manning's equation to obtain estimates of the velocities at the simulated discharge:

$$V_i' = \frac{1.49}{n_i} d_i^{2/3} S^{1/2} \quad (25)$$

where all terms were defined for equation 24.

Equation 25 provides an initial estimate of the velocity at a vertical, V_i'. The final estimate of V_i is determined through a mass balancing function performed internally in IFG4:

1) A simulation discharge (Q_{sim}) is specified as input to the model.
2) First-order estimates of depths (d_i') and velocities (V_i') are produced for all verticals and a width (w_i') is also determined for newly wetted cells).
3) An interim discharge (Q_{temp}) is calculated from these first estimates of the hydraulic variables:

$$Q_{temp} = \sum_{i=0}^{n} w_i' d_i' v_i' \quad (26)$$

Q_{sim} and Q_{temp} will probably not be the same because Q_{temp} is based on predicted values that contain error. The farther Q_{sim} is from the calibration discharge, the greater the aggregate error in Q_{temp}.

4) In order to make Q_{sim} and Q_{temp} come out the same (i.e., mass balance), IFG4 calculates a velocity adjustment factor (*VAF*):

$$VAF = \frac{Q_{sim}}{Q_{temp}} \quad (27)$$

5) The final velocities that are forwarded to the microhabitat simulation portion of PHABSIM are determined by:

$$V_i = (V_i')(VAF) \quad (28)$$

Error Analysis

The *VAF* is one of the most important quality indicators in PHABSIM's hydraulic component. Recall that in IFG4, the calibration velocities are used to calculate values for Manning's n. Once these n-values have been entered, they are constant as far as IFG4 is concerned. Thus, when simulated discharges are higher than the calibration discharge, the true value of Manning's n should be smaller than the calibrated n-values (Fig. 4-34). Similarly, when lower flows are simulated, the calibrated n-values will be smaller than they should be.

If the calibrated values of n are too high, the predicted velocities will be too small, and Q_{temp} will be smaller than Q_{sim}. This combination of factors will cause the *VAF* to be larger than 1.0 by a factor sufficient to make Q_{temp} equal to Q_{sim}. Just the opposite effect happens if the calibration n-values are lower than they should be for a simulated discharge. If the hydraulic simulation component (regardless of what combination of models is used) is performing according to hydraulic theory, *a plot of VAFs versus discharge should look like an inverted image of the Manning's n - discharge relationship* (Fig. 4-35).

Because of its theoretical underpinnings, the *VAF* plot is one of the quickest and easiest tools for assessing the quality of hydraulic simulations. As a rule-of-thumb, the hydraulic component is performing as expected if the VAF plot looks like the one in Fig. 4-35. The range of VAFs should mirror the range of Manning's n that one would obtain if WSP were calibrated to the extreme simulated discharges. For example, suppose the values for Manning's n used to simulate the high and low water surface elevations were 0.025 and 0.100, respectively, Manning's n at the velocity calibration discharge was 0.040, and the VAF at the calibration flow was 1.0. Since the low-flow value of Manning's n was 2.5 times the value at the velocity calibration flow, it would be reasonable to expect a *VAF* of about 2.5 for the low-flow simulation. By similar logic, one could expect *VAF*s for the high-flow simulation to be in the neighborhood of about 0.63 (0.025/0.040).

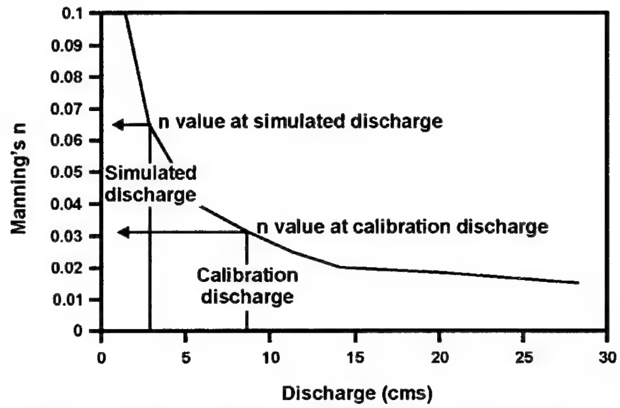

Fig. 4-34. Manning's n from calibration compared with Manning's n at a simulated discharge.

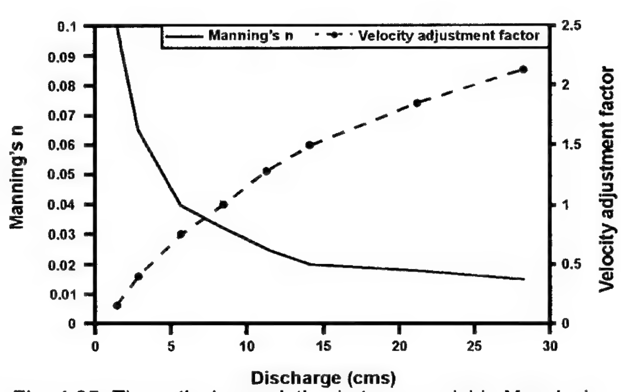

Fig. 4-35. Theoretical association between variable Manning's n and the velocity adjustment factor in IFG4.

VAF plots that differ substantially from Fig. 4-35 may be symptomatic of unusual or erroneous relationships between water surface elevation and discharge. Errors in water surface elevations show up so graphically in *VAF* plots because depth is derived from the water surface elevation and is used twice in the mass balance equation:

$$Q = \sum_{i=0}^{n} (w_i)(d_i)(v_i) = \sum_{i=0}^{n} w_i d_i \frac{1.49}{n_i} d_i^{2/3} \sqrt{S_e} \qquad (29)$$

VAF plots are particularly useful in identifying oversteepened rating curves, which result in estimated water surface elevations that are too low at low flow and too high at high flow. If the water surface elevation is too low, the depths and velocities across the transect will also be too low for the simulated discharge. Consequently Q_{temp} will be smaller than Q_{sim} and the *VAF* will be greater than 1.0. Just the opposite happens at the high flows, where the predicted depths will be too large. The net result is a *VAF* plot that is the exact opposite of its expected theoretical distribution.

Although *VAF* plots that differ from Fig. 4-35 usually indicate a problem with the stage-discharge relationship, it seems as though nothing occurs in PHABSIM without at least one exception to every rule. The exception is the effect of a variable backwater. A variable backwater occurs when a backwater is present at high flows but absent at low flows. This phenomenon commonly occurs when a hydraulic control is inundated at high flows by the backwater of another control farther downstream. The effect of the variable backwater on the *VAF* plot depends on whether the backwater was present or absent when the calibration velocities were measured (Fig. 4-36).

Another useful indicator of model performance is a longitudinal plot of the stream's thalweg and water surface profiles (Fig. 4-37). The water surface profile should not have any bumps, dips, or other features suggesting that water runs uphill anywhere along the longitudinal profile. (A popular tenet of hydrology is that water only runs uphill towards money.) The profile should be relatively steeper in riffles and flatter in pools and should follow streambed irregularities more closely at low flows than at high flows. Profile analysis is based more on common sense than on theory, but it can be used in conjunction with *VAF* analysis. If something looks suspicious along the longitudinal profile, it may be possible to confirm or reinforce your misgivings by examining the *VAF* plot.

Microhabitat Simulation Options

The algorithms for calculating microhabitat area given in equations 11 and 12 are the standard defaults for PHABSIM. Following up on the discussion of habitat suitability criteria, however, the default models might not be appropriate for the microhabitat attributes most important to the target species. It should not come as a surprise, therefore, that it may be necessary to use an alternative approach in the calculation of weighted usable area. (Note that if binary criteria are used, the output from PHABSIM is a real area, not weighted usable area.) Although many of the more important alternative algorithms are data-independent (different calculations of microhabitat area can be performed using the same basic set of data), some will require additional or different data.

Nose Velocities

One option we have already mentioned several times is the use of nose velocities rather than mean column velocities in the calculation of the composite suitability index. The use of the nose velocity may be more important in large, deep rivers than in small streams, because in deep streams the near-bottom velocity can be substantially

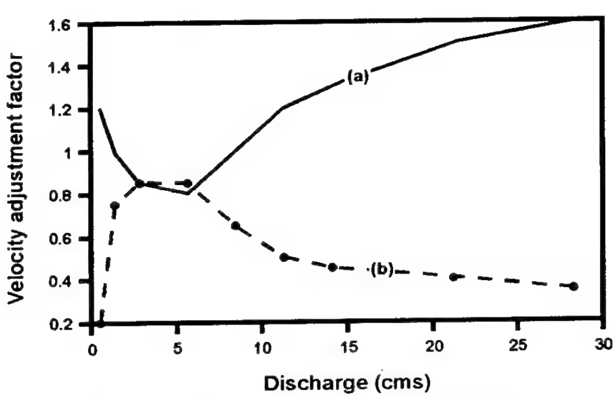

Fig. 4-36. Velocity adjustment factor (VAF) versus discharge plots that can occur in stream sections subject to variable backwaters. In (a) the backwater was present when calibration velocities were measured, while in (b) backwater was absent when calibration velocities were measured.

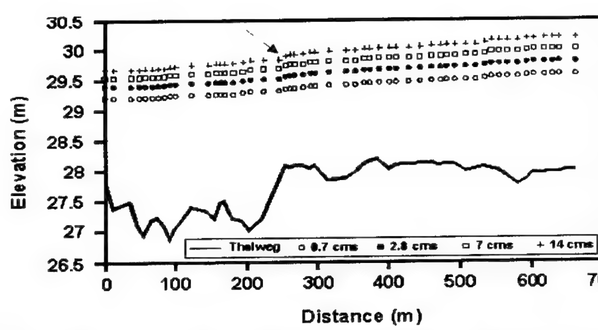

Fig. 4-37. Longitudinal profiles of the thalweg and water surface elevations associated with different simulated discharges. The arrow points to a potentially oversteepened portion of the profile that might suggest a problem with simulations over 14 cms. One should check the high flow calibration data to determine if this jump in the water surface is real or an artifact of the hydraulic model.

slower than the mean column velocity. The use of the mean column velocity in a large stream (especially if the criteria came from a small one) may result in unrealistically low estimates of microhabitat. In PHABSIM there are several choices for modeling nose velocities. The easiest option (i.e., the one requiring the least empirical data) is known as the 1/7th power law (Milhous et al. 1989), in which the nose velocity is calculated as:

$$V_n = 1.143\, V_{mc} \left(\frac{d_n}{d}\right)^{1/7} \tag{30}$$

where V_n is the nose velocity, V_{mc} is the mean column velocity, d_n is the depth at which the nose velocity is to be calculated (nose depth), and d is the depth of the water column.

At the other end of the empirical data spectrum, the user has the option of defining the constant and exponent terms of equation 30. In this case, a representative sampling of nose and mean column velocities and depths are collected, and regression analysis used to determine terms a and B in equation 31:

$$\frac{V_n}{V_{mc}} = a \left(\frac{d_n}{d}\right)^{B} \tag{31}$$

Other options include determining the nose velocity using a form of the Prandtl-vonKarman universal velocity distribution law, shear stress, or Froude number (Chow 1959). A cautionary note is warranted with respect to nose velocity equations, no matter which one you select. Nose velocities are "second-generation" simulations within PHABSIM. They are derived from predictions of mean column velocities. This means that regardless of the quality of the simulated mean column velocities, the predictions of nose velocities can only be less accurate. Furthermore, as the substrate size becomes large relative to the nose depth, the accuracy of the nose velocity predictions will usually deteriorate. A rule-of-thumb is to set the nose depth no less than about twice the average size of the bed materials. In other words, if the bed is composed mostly of 15-cm cobbles, the nose depth should probably be about 30 cm above the streambed.

Effective Habitat Using HABEF

The HABEF program (Milhous 1991) was developed to examine the effects of fluctuating streamflow levels on microhabitat availability for organisms having limited mobility. The underlying concept of HABEF is that under unsteady flow conditions, a stream cell is suitable for organisms of limited mobility only if it is suitable throughout the range of flows experienced by the organism (Fig. 4-38). Effective habitat is calculated for each cell by comparing the suitabilities of the cell at each of two streamflows. In the overall comparison of the two discharges, some cells will be more suitable at high flows and some at low flows; HABEF records the lower of the two paired values for the cell. The effective composite suitability is then multiplied by the cell's surface area for the calculation of weighted usable area. Although HABEF was first developed to quantify the effects of unsteady flow on salmonid redds, it has since been used to evaluate the impacts of hydropeaking on young fish and aquatic macroinvertebrates (Bovee 1985).

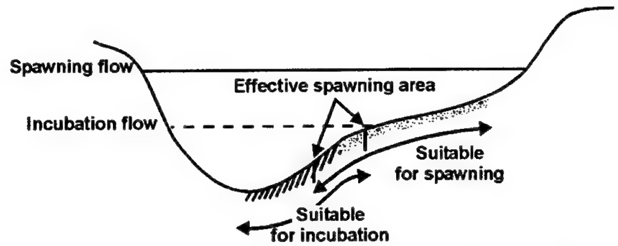

Fig. 4-38. Depiction of the effective habitat concept as it relates to spawning and incubation in a temporally varied streamflow regime. The area suitable for spawning at a high discharge is shown with a stipple pattern. The area that was suitable for incubation at a lower discharge is delineated by cross-hatching. The effective spawning habitat is indicated by the area where the two patterns overlap.

Feeding Stations Using HABTAV

The HABTAV program was designed to simulate the combination of habitat features that provide high-energy feeding stations for drift-feeding fish. These feeding stations consist of a low-energy holding area in close proximity to a high-energy drift delivery area (Fausch 1984). HABTAV calculates the suitability of a cell using a combination of an occupied velocity (either mean column or nose) and an adjacent velocity. The adjacent velocity criteria represents velocities in nearby cells (or in the same cell) that are desirable for the delivery of drifting food items. To use this program, habitat suitability criteria are necessary to describe the suitable range of adjacent velocities and an appropriate lateral search distance for feeding stations. In practice, adjacent velocity criteria and search distances are rarely available, so these criteria are frequently developed using category I techniques.

Integrating Macrohabitat and Microhabitat

Alternatives are analyzed in IFIM through the generation and evaluation of habitat time series. At the smallest geographic scale, these time series are based on the total amount of habitat available for a target species in a segment. The operative phrase here is total habitat, because up to this point we have only generated discharge relations with macrohabitat and microhabitat. In order to make our information usable for the next phase of IFIM, it is necessary to combine output from the macro- and microhabitat components into a single relationship of total habitat versus discharge. In the following sections, we

will discuss two methods of integrating microhabitat and macrohabitat in IFIM studies: binary and numerical integration.

Binary Integration

Binary integration is the simplest and most straightforward method of combining macrohabitat and microhabitat relations. Under this approach, the macrohabitat suitability criteria are in binary format, that is, either suitable or unsuitable. When binary criteria are used, the output from the macrohabitat component of IFIM is the suitable length of stream for the target species at each simulated discharge. Output from the microhabitat component is in units of microhabitat area per unit length of stream. Thus, for a given streamflow, total habitat area is calculated by:

$$HA\,(m^2) = SL(km)\;WUA\,(m^2/km) \qquad \textbf{(32)}$$

where *HA* is the total habitat area in the segment for the target species at discharge *Q*, *SL* is the length of stream having suitable water quality and temperature for the target species at discharge *Q*, and *WUA* is the unit of microhabitat area for the target species at discharge *Q*.

Binary integration is only a little more complicated when habitat typing is used to describe microhabitat distributions. *WUA*, as expressed in equation 32, should be calculated as a weighted mean for all of the PHABSIM sites in the segment prior to integration with macrohabitat:

$$WUA_{(all)} = (w_1)(WUA_1) + (w_2)(WUA_2) + \ldots (w_n)(WUA_n) \qquad \textbf{(33)}$$

where $WUA_{(all)}$ is the weighted average unit microhabitat area for the entire segment, w_i is the proportion of mesohabitat type *i* in the segment, and WUA_i is the unit microhabitat area for mesohabitat type *i*.

Numerical Integration

The difference between numerical and binary integration is the format of the macrohabitat suitability criteria for the target species. In contrast to binary criteria, which have suitability values of 0 or 1, numerical integration uses criteria in univariate curve format. With this format, suitability values vary between 0 and 1. This sliding scale allows the use of multifaceted criteria. For example, a broad range of temperatures (e.g., from 7° to 26.5° C) might be suitable for survival during the growing season, but a narrower range (e.g., from 11° to 20° C) promotes the highest growth rate in the target species. This "two-stage" definition of temperature criteria results in a temperature suitability curve (Fig. 4-39). Suitabilities for temperatures between survival and growth limits are estimated by interpolation.

To perform a numerical integration of macrohabitat and microhabitat for a given discharge, the temperature for an incremental length of stream (e.g., 1 km) is read from the

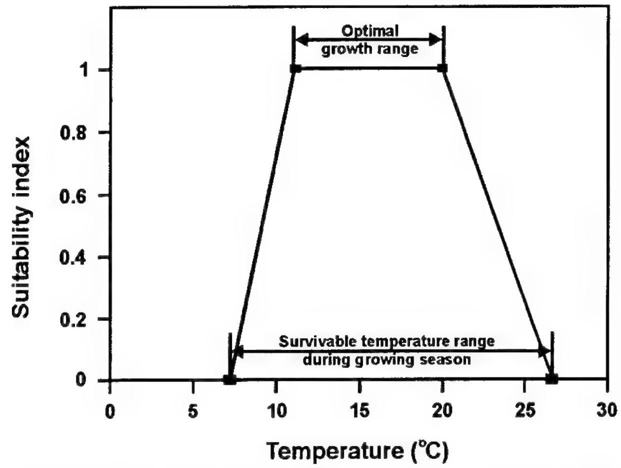

Fig. 4-39. Temperature suitability index curve constructed by overlaying two sets of binary criteria for growth and survival. The suitability index is a sliding scale wherein a score of 1.0 represents the optimum temperature range and 0.0 indicates unsuitable temperatures.

thermal profile. The temperature suitability, interpolated from the temperature criteria curve, is used to calculate the total (temperature-conditioned) habitat for the incremental length of stream:

$$HA_{(Q,i)} = \sum_{i=0}^{n} (WUA_{(Q)})\,(SI_{Q,i})\,(L_i) \qquad \textbf{(34)}$$

where $HA_{(Q,i)}$ is the total habitat in increment (*i*) at discharge (*Q*), $WUA_{(Q)}$ is the unit microhabitat for the segment at discharge (*Q*), $SI_{(Q,i)}$ is the temperature suitability for increment (*i*) at discharge (*Q*), and $L_{(i)}$ is the length of increment (*i*).

After solving equation 34 to find the first incremental length of the segment, proceed to the next increment and repeat the procedure. The total habitat for the entire segment is calculated by summing $HA_{(Q,i)}$ across all of the incremental lengths of the segment. The procedure is repeated for every discharge for which total habitat information is needed.

Seasonal Considerations

From a mathematical standpoint, there is nothing particularly complicated about either form of habitat integration. However, one must be careful not to oversimplify real-world phenomena by assuming that the same discharge produces the same amount of habitat for a species or life stage all the time. If this statement seems counterintuitive, consider the idea that the same streamflow will produce different suitable lengths of stream during different times of the year. Differences occur because of seasonal changes in the factors affecting temperature and water quality (e.g.,

thermal loading, shading, waste loading, and assimilation capacity). At a given discharge, therefore, temperature and water quality will not be the same throughout the segment all the time. To realistically depict seasonal variations in macrohabitat, it may be necessary to develop separate functional relationships between discharge and suitable stream length for each season of the year.

Less obvious are seasonal differences that occur between discharge and unit microhabitat for a life stage. The same discharge does not produce the same amount of microhabitat at all times because the behavior of a species commonly changes over time. Growth of individuals, shifts in feeding behavior, changes in activity for migration, reproduction, or winter dormancy are all manifested as shifts in microhabitat preferences (Bovee et al. 1994). If changes in behavior result in detectable shifts in microhabitat use, it may be necessary to use seasonally explicit habitat suitability criteria to quantify microhabitat. To realistically depict microhabitat for rapidly growing young-of-the-year fish, for example, it may be necessary to use different microhabitat suitability criteria for each month during the first growing season.

Suggested Reference Materials

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Buchanan, T. J., and W. P. Somers. 1969. Discharge measurements at gaging stations. Techniques of Water-Resources Investigations of the United States Geological Survey, Book 3, Chapter A8. U.S. Geological Survey, Washington, D.C. 65 pp.

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Chapter 5. IFIM Phase IV

Alternatives Analysis and Problem Resolution

As we enter the final phase of an IFIM analysis, several philosophical concepts warrant repeating:

- The reason that you are involved in an IFIM analysis in the first place is because some entity (perhaps your own agency) has proposed an action that will change the habitat characteristics of the stream under investigation. Your primary responsibility is to address the issues and impacts associated with the proposed action.
- You are involved in an incremental problem, not standard-setting (see Stalnaker et al. [1995] if you are still uncertain about the difference). This chapter deals with preparations for negotiating proposed actions and alternatives.
- The currency for evaluating alternatives in virtually all IFIM studies is total habitat, not microhabitat, numbers of fish, or money.
- IFIM is not designed to produce the "one best answer." The best answer is whatever the consensus of the stakeholders says it is.

It should come as no surprise that by the time you reach this point in most IFIM studies, a considerable amount of negotiation has already taken place: the objectives and scope of the study plan, the layout of study sites, which simulation options to use, and what habitat metrics to include in the analysis are all negotiable during an application of IFIM. We have observed that groups that start negotiating during the early phases of the study usually have an easier time during the latter phases.

Alternatives analysis cannot really be separated from problem resolution, because both are parts of an iterative problem-solving cycle. Not surprisingly, the solution technique on which IFIM is based is a form of incrementalism. Negotiated settlements result from a repetitive process by which (1) an alternative is proposed, (2) the effects of the alternative are measured and evaluated, and (3) improvements on the alternative are proposed, tested, and negotiated. Eventually, this process leads to one of two outcomes. Either a mutually agreeable solution will emerge or the negotiation will reach an impasse. If an impasse cannot be overcome, the responsibility for making the decision is passed on to a higher authority (i.e., an arbitrator).

Information is a source of power in negotiations, but mere possession of information is not as powerful as the ability to use it in support of an objective. Previous chapters have discussed how to accumulate information needed to solve a problem. In this chapter, we describe how to leverage the power of information obtained from IFIM through an iterative process of formulating and testing alternatives. Specifically, we will describe tools and processes to help you articulate an alternative and to evaluate its effectiveness, feasibility, and associated risks. We also introduce some basic concepts of natural resource negotiation, various negotiating strategies, and some of the tactics that you might be exposed to during a negotiation.

Preparing for Negotiation

Finding Your BATNA

In an application of IFIM, we expect to resolve problems by negotiation, whether in the distributive political arena or the regulatory arena. Despite our best intentions, however, we sometimes end up in situations where negotiating seems impossible. For example, it may become apparent after a few sessions that one of the stakeholder groups has no intention of negotiating anything. In the movie *Sometimes a Great Notion*, Henry Fonda summarized this kind of negotiating stance in the phrase "never give an inch." When dealing with adherents to this philosophy, the term "negotiation" is a misnomer because the stakeholders will not compromise or even feign bargaining in good faith. They are willing to accept any concessions your side makes but will not offer any of their own. BATNA stands for Best Alternative To a Negotiated Agreement. The purpose of the BATNA is to help us decide when it is time to quit trying to negotiate and start pursuing other courses of problem resolution, such as litigation.

Prior to entering the first negotiation session (or better yet, prior to starting the study), you should discuss with your colleagues and constituents what the most likely outcome would be for your interests if the negotiation were to fail. Often, judicial precedent is a good place to look for clues to your BATNA, because arbitration is the logical counterpoint to negotiation. How have the courts decided on issues that were similar to this one? How have FERC administrative law judges ruled in similar cases? If you learned that the courts have ruled in favor of "your side" 90% of the time, you might be less inclined to stick with a difficult negotiation. Conversely, you might decide that no matter how difficult the negotiation, your interests would be better served than if the case went to litigation.

Positional Bargaining

When individual negotiators attempt to protect or enhance their own objectives exclusively, they are practicing what is known as positional bargaining. The goal of positional bargaining is simply to protect one's position whenever it is threatened by an alternate proposal. Sometimes, positions are protected even when they are not threatened, simply to avoid the appearance of giving in to the opposition. Despite its widespread use, there are two major drawbacks to positional bargaining. First, positions often

become targets. Instead of recognizing the legitimacy of an issue or concern, the opposition attempts to impugn its importance. Second, attacks on positions may become personalized. This can lead to retaliation, resulting in a mean-spirited negotiation oriented toward the negotiators instead of toward the problem.

Idealized Objectives

Alternatives are guided by the goals and objectives of their proponents. A water management alternative proposed by an irrigation district will undoubtedly look very different from one offered by a fisheries resource agency, at least initially. The goal of the irrigation district might be to maximize corn production, whereas the goals of the fisheries resource agency may be to maximize sport fish production and recreational opportunities. When formulating alternatives, it is important to distinguish between an objective and the means by which the objective can be achieved. For example, water allocation is a common objective to the participants of a negotiation, but water allocation can also be considered as a means of achieving an objective. If your objective is to improve fish habitat, remember that changing the streamflow may be one of several ways to accomplish the goal.

The idealized objective is a problem-solving device designed to help stakeholders move away from positional bargaining and toward integrative problem-solving. In concept, the purpose of an idealized objective is to consolidate the disparate goals of the negotiating parties. An idealized objective can be constructed by asking the simple question: "What would the perfect solution to this problem look like?" Using an irrigation district, fisheries agency, and a group of homeowners around a reservoir as an example, an idealized objective might be to develop a release schedule for the reservoir that will guarantee full delivery of the authorized firm yield to the irrigation district, that will restore instream habitat for rainbow trout and brown trout to predevelopment conditions, and that will result in no more than 1 m of drawdown during summer recreation periods.

The development of an idealized objective may require the services of a facilitator or other neutral party, because negotiators typically enter the bargaining arena with a positional mindset. To recognize the legitimacy of another's issues or concerns may be interpreted as a threat to one's own position. Overcoming mistrust and insecurity is a major hurdle in designing an idealized objective. The intent of an idealized objective is to make the goals of individuals the common goals of the group. When committed to by the group, an idealized objective can be a powerful tool for innovative problem solving.

Focused Biological Objectives

How an objective is stated depends on the nature of the proposed change, the institutional and decision-making arena, and the availability of biological information. For example, the biological objective for the preservation of an existing fish community is often no net loss of habitat. Loosely translated, this means that the amount of habitat available for all life stages of all target species must be the same after the project is in operation as it was before the project was built. In studies involving the restoration of previously altered streams, the biological objective may be to approximate the amount of habitat available for each life stage and species under preproject conditions. This objective, as applied to restoration, has sometimes been termed historical mitigation, because the goal is to mitigate project impacts that first occurred long ago. Occasionally, the objective of the fisheries resource agency is to optimize habitat resources for a few high-profile target species. Although flow regimes to optimize habitat can be determined with IFIM, such an alternative may be viable only under very limited circumstances (such as when the primary goal of the reservoir operator is to optimize the downstream fishery).

The "no net loss" alternative can be formulated and tested with little or no biological data. The only information necessary to test this alternative is the historical availability of habitat; then, alternatives are developed to match the pattern of historical availability as closely as possible. Many stakeholders will argue against this approach on the grounds that it reduces their flexibility and options and that providing nonessential habitat for unimportant life stages is a waste of resources.

Regardless of these arguments, the fisheries resource agency may have little choice but to adopt a "no net loss" or "historical mitigation" objective if biological information is totally lacking. With no data on the population dynamics of the target species, it is difficult to figure out which are the most important life stages and types of habitat. Consequently, the safe option is to protect all of them. One way to overcome this problem is to identify the need for this information during the problem identification phase and include the collection of biological data in the study plan.

Where biological data are available, it may be possible to identify critical habitat types and habitat bottlenecks. A critical habitat type is one that is known to be important to the well-being of a species or group of species. Examples of habitat types that have been so designated in past applications of IFIM include effective spawning habitat, feeding stations for drift feeding fish, riffles for the production of aquatic macroinvertebrates, and quiet backwaters as rearing areas for larval fish.

Some habitat bottlenecks are fairly obvious, whereas others are subtle. Some are related to the magnitude and timing of short-term events, others affect populations over longer periods. Knowledge of critical habitat types and bottlenecks can be extremely advantageous in formulating alternatives. If these constraints can be identified, the habitat objective can be focused on alleviating the limitation.

The most valuable biological information for ferreting out habitat bottlenecks is a cohort table, constructed from age, growth, and population data collected over a series of 5-10 years. Cohort tables (Table 5-1) allow investigators to determine the number of young fish produced each year (year-class strength) and then track the fate of individual year classes with increasing age. The example presented in Table 5-1 suggests that carryover of year classes was relatively strong, especially from age 0 to yearlings. Where such a response to recruitment is evident, juvenile and adult populations will often be associated with habitat conditions that affect early life history phases. For example, spawning habitat, fry rearing habitat, or second-order effects of thermal regimes on first-year growth rates may be the most important determinants of adult population size (Nehring and Anderson 1993; Bovee et al. 1994). If there is little evidence of carryover in year classes, populations of juveniles and adults may be near carrying capacity, implying a stronger relationship with adult habitat or the habitat for prey species, competitors, or predators. Where populations are near carrying capacity, there will often be a good correspondence between the amounts of juvenile or adult habitat and population sizes from year to year.

Unfortunately, cohort tables take a long time to assemble. If you do not already have a cohort table well underway (i.e., 3-4 years completed) by the time you start an IFIM study, you are not likely to have one in time for alternatives analysis. It may be possible, however, to identify potential habitat bottlenecks from limited biological data and some educated guesswork. Here are some characteristics of the population to look for:

1) Are adults of the species heavily harvested? Harvest is the equivalent of selective predation on adults. If a population is fished heavily, it is probably below carrying capacity and will likely be responsive to changes in recruitment. In these populations, look for critical habitats and events associated with early life history to emerge as bottlenecks. Conversely, populations protected by catch-and-release or similar restrictions may be closer to carrying capacity and their numbers more likely to be related to adult habitat.
2) Is the population short-lived? Short-lived populations imply the necessity to maintain a steady supply of recruits to maintain the adult population.
3) Does the population have a normal-looking age structure? A real key here is to look for abnormally strong, weak, or missing year classes. Obviously, a population with a life span of 20 years that is dominated by 19-year-old fish is in dire need of recruitment.
4) Does the population exhibit good growth rates and condition? If not, the thermal regime may be a problem, or food producing habitat may be in critically short supply during all or parts of the year.
5) Is there evidence that survival rates are related to growth or condition? You may be able to find the answer in the literature, but growth and survival may depend on the species and geographic location. For example, Shuter and Post (1991) found that survival to age 1 in smallmouth bass was strongly related to their size at age 0 in the northern United States and southern Canada. Sabo (1993) found no such relation in smallmouth bass populations in Virginia.

Table 5-1. Example of a cohort table for smallmouth bass from the Huron River, Michigan, from Bovee et al. (1994).

Year class	Number at age 0	Number at age 1	Number at age 2	Length at age 0 (mm)
1982			78	
1983		63	50	
1984	67	54	14	76.4
1985	585	105	69	89.0
1986	267	78	50	93.0
1987	1,318	217	81	100.8
1988	499	94		89.2
1989	88			83.8

Test As You Go

Some alternatives that emerge from brainstorming sessions may initially seem infeasible, unrealistic, or just plain silly. What initially seem to be outlandish proposals, however, sometimes turn into the elegant solutions discussed by Fisher and Ury (1981). There may be a tendency to dismiss alternatives before they are fully developed or without a thorough evaluation. Testing each alternative as it is proposed minimizes the amount of time spent "negotiating over unknowns," which occurs when negotiators incorrectly presume that they know how an alternative will turn out in the analysis. In the true spirit of incrementalism, the identification of alternatives that produce the desired results is an important step in suggesting improved versions. By taking many small, positive steps, the negotiating team may settle on a solution that would have been dismissed outright at the beginning of the negotiation.

How to Test Alternatives

In addition to being mutually acceptable among stakeholders, a good alternative should be effective, feasible, and sufficiently flexible to accommodate the risk of failure. Effectiveness is a measure of how well an alternative meets a biological objective. Feasibility determines whether an alternative can actually be implemented. Risk analyses are conducted to determine how often and under what circumstances an alternative will fail. Risk analysis in IFIM studies usually involves the formulation and testing of

contingency plans to be enacted when the primary alternative fails.

Effectiveness

Habitat Time Series

Under its most basic definition, effectiveness of an alternative is determined by comparing the amount of habitat available under the alternative with the habitat available under the baseline. There are several variations to this theme, and different measures of habitat availability can be used to interpret effectiveness. However, the basic tool for quantifying differences between baseline conditions and an alternative is a habitat time series. The procedures for constructing a habitat time series are:

1) From the hydrologic time series, find the discharge for the initial time-step (Fig. 5-1a).
2) From the discharge-total habitat relationship for the life stage or species, find the habitat area corresponding to the discharge from step 1 (Fig. 5-1b).
3) Copy the habitat area into the position of the first time-step in the habitat time series (Fig. 5-1c).
4) Repeat the process for all of the time-steps in the hydrologic time series.

There are two basic ways to construct habitat time series in IFIM. The first option is to use the programs available in the IFIM time series library, TSLIB (Milhous et al. 1990). The second approach is to program your own habitat time series in your favorite computer language or use commercial spreadsheet software. Experienced users of IFIM tend to prefer to develop their own time series, but spreadsheets are popular because of their flexibility and superior graphics capabilities.

Regardless of how habitat time series alternatives are produced, they can include several ways to achieve an objective:

1) Vary the flow regime. The input for this comparison consists of two hydrologic time series (baseline and alternative) and a single discharge-habitat relation for a life stage. This arrangement is the one most commonly used in IFIM applications.
2) Vary microhabitat-discharge relations. The input for this option is a single hydrologic time series to represent both baseline and postproject conditions and two or more discharge-microhabitat relations. This selection could be used to assess stream channelization or habitat improvement projects.
3) Vary macrohabitat-discharge relations. In this case, input is manipulated to reflect management options to affect habitat suitability in a longitudinal fashion. For example, one might reduce loading rates to represent improved wastewater treatment or modify the streamside shading to simulate protection of a riparian corridor. The list of options can be quite extensive, especially if baseline macrohabitat conditions are marginal.

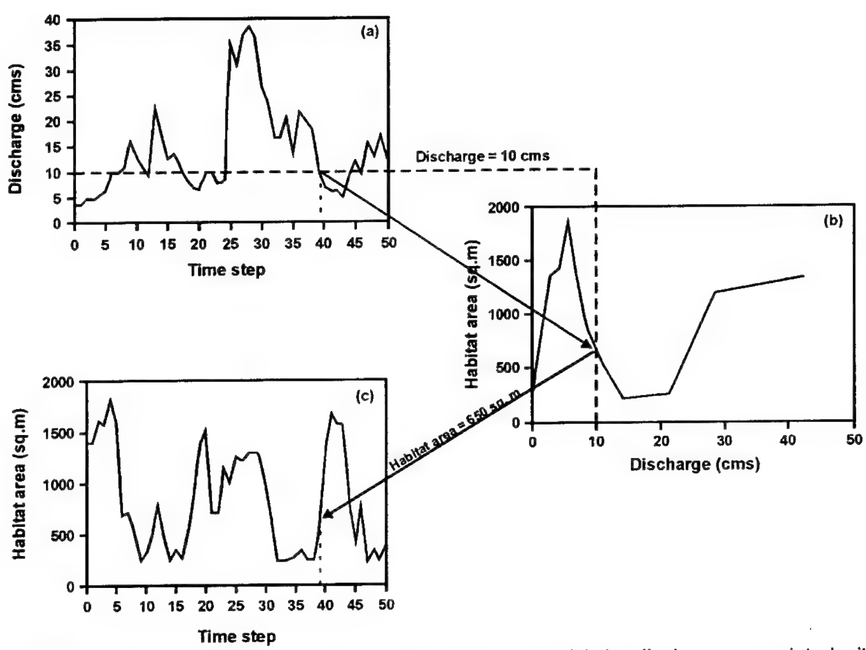

Fig. 5-1. Ingredients for constructing a habitat time series: (a) the discharge associated with a time-step is read from the hydrologic time series, (b) the total habitat area for the selected discharge is obtained from the discharge-habitat relationship, and (c) the total habitat area for the time step is entered into the habitat time series.

4) Do everything. This approach might examine the combined effects of modifying the flow regime and altering both instream microhabitat and macrohabitat. Although more complex in design, this option allows the greatest flexibility in the analysis of alternatives.

Habitat time series can be displayed in either graphical or tabular form. Attributes of the habitat time series are similar to those of a hydrologic time series. Both types of time series preserve the chronological ordering of events and can provide comparative information between baseline and alternative conditions. The nonlinear relation between discharge and habitat (e.g., Fig. 5-1b), however, can complicate interpretation of the habitat time series. The lowest values in the habitat time series regularly occur at the extreme high and low discharges, whereas the largest amounts of habitat are available at intermediate discharges. Consequently, the habitat time series often look nothing like the hydrologic time series used to generate them (compare Fig. 5-1a with 5-1c, for example). Although most people acknowledge that the habitat-flow dynamics occur just as we have described them, the relationships between the hydrologic and habitat time series are, nonetheless, sometimes counterintuitive.

Habitat Duration Curves and Metrics

Superimposing the habitat and hydrologic time series provides information about the types of flow events that cause habitat reductions at different times of the year. This information can be particularly useful in the identification of potential habitat bottlenecks, especially if it is accompanied by a modicum of biological data. It is very difficult, however, to quantify differences in habitat availability from a habitat time series. Quantification of habitat is made considerably easier through the use of a habitat duration curve (Fig. 5-2). A habitat duration curve is constructed in exactly the same way as a flow duration curve but uses habitat values instead of discharges as the ordered data.

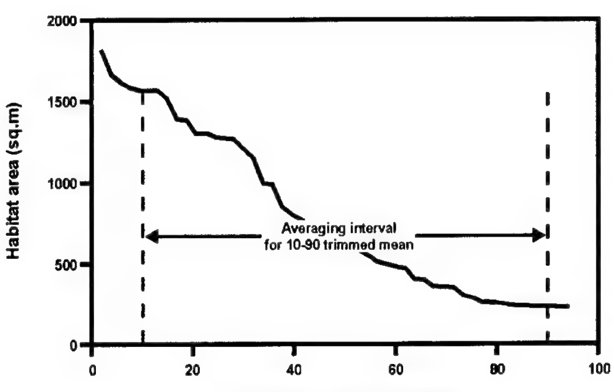

Fig. 5-2. Averaging interval for a 10-90% exceedance trimmed mean from a habitat duration curve.

Although habitat duration curves look like flow duration curves, there is no direct correspondence between the two. For example, the habitat value that is exceeded 90% of the time usually does not correspond to the discharge that has the same exceedance probability. This discordance happens because of the aforementioned bell-shaped relationship between total habitat and discharge. The same amount of habitat can occur at two or more different discharges. Consequently, some confusion can arise from reading habitat duration curves, because a habitat area with a given exceedance probability might be related to several discharges (all having different probabilities of exceedance). Therefore, we suggest that the habitat time series be used to determine where problem areas exist and how they are related to discharge; the habitat duration curve should be used to quantify the differences in habitat between baseline and alternative conditions.

The habitat duration curve is valuable for quantifying differences in habitat availability because a variety of habitat metrics can be extracted from the ordered data. Most habitat metrics in IFIM are derived by averaging over various portions of the duration curve. The relevance of any particular metric depends on the mechanisms that create habitat bottlenecks for the target populations.

The most common and easily understood habitat metric is the average of all habitat events in the time series. The trimmed mean (Fig. 5-2) is a variation that excludes the extreme highs and lows in the series. In Fig. 5-2, values with exceedance probabilities less than 10% and greater than 90% were excluded from the mean. Trimmed means are used primarily when whole-series averages are skewed by extreme events. Whole-series averages and broadly defined trimmed means imply that extreme, rare events are not considered to be very important biologically.

Another biological implication of the whole-series and trimmed means is that periods of abundant habitat can offset periods of restricted habitat. This implication deserves serious thought. Habitat bottlenecks affecting adult fish appear to result from overcrowding (Bovee 1988). Density-dependent effects such as agonistic behavior, reduced growth or condition, and increased transmission of disease eventually affect the adult survival rates. The chronic nature of adult bottlenecks suggests that the important habitat characteristic may be the aggregate of all of the events in the time series, or of only the lowest ones, depending on their sequence in the time series. For example, a population that is alternately compressed and allowed to expand every other week might not respond the same way as a population that was compressed for 6 months consecutively. In the first case, the whole series average or a broadly trimmed mean might be the best way to assess changes in habitat. The second case could be represented either by taking the average of the lowest 3-6 consecutive months in the time series or by using a special case of the

trimmed mean. The special case consists of trimming all of the habitat values above the median and averaging the habitat values between the median and the minimum (Fig. 5-3). This averaging interval is equivalent to counting only the "troughs" in the habitat time series. Taking the average of a certain number of consecutive months is probably a better approach if the habitat limitation tends to be persistent and consistent over a portion of the year. The special trimmed mean may be more appropriate if extended periods of limited habitat are scattered throughout the year.

Habitat minima, 90% exceedance values, or other events having high exceedance probabilities can all be used to quantify extreme, low-frequency habitat events. These metrics have been associated with survival rates of early life history phases of fish (Nehring and Anderson 1993; Bovee et al. 1994). One needs to be careful in using these metrics as impact assessment devices, however, because they may fail to quantify some kinds of habitat changes. For example, the 80% or 90% exceedance values may not detect changes in magnitude of the habitat minima. Likewise, the minimum value will not quantify changes in frequency of other low events in the series (Fig. 5-4). In order to depict both types of change in the habitat duration metric, we suggest using an average of the lowest habitat events (e.g., average from 80%-100% exceedance probabilities).

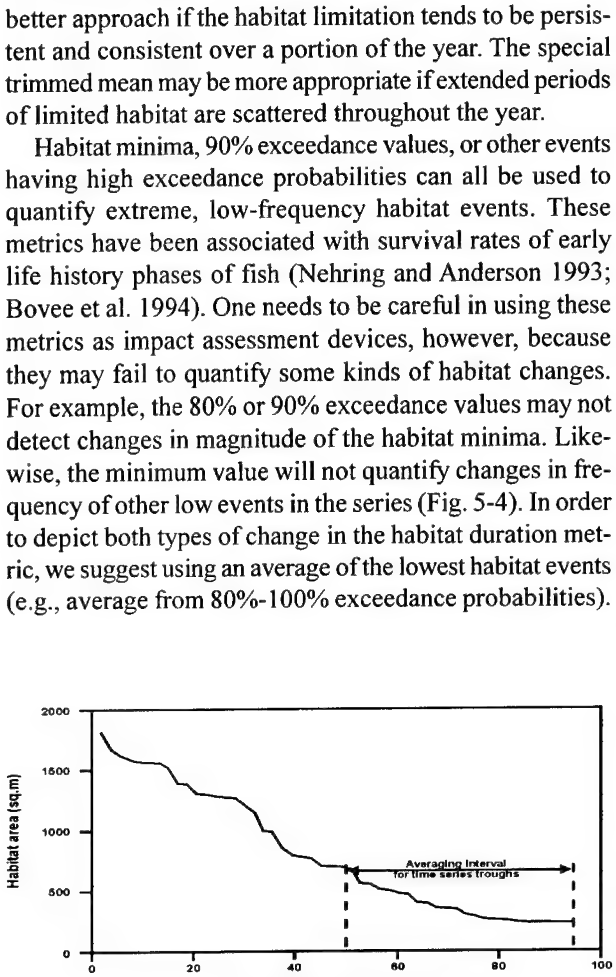

Fig. 5-3. Averaging interval for a 50-100% trimmed mean from a habitat duration curve. A habitat time series can be considered as a series of peaks and troughs in habitat availability. This metric is the average of all the troughs.

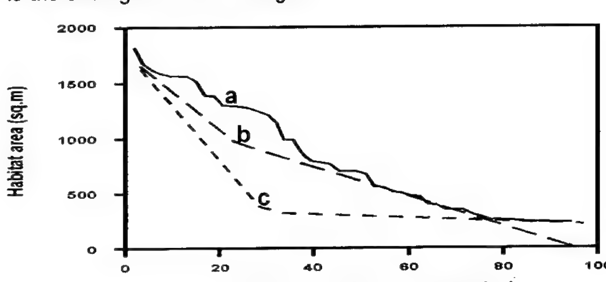

Fig. 5-4. Limitations of fixed exceedance value metrics in assessing impacts to acute, short-term habitat bottlenecks: (a) baseline habitat duration curve; (b) 80% exceedance value same as baseline, but minimum is lower; and (c) minimum value is same as baseline, but frequency has increased.

When this approach is used, any change in short-term habitat minima, either magnitude or frequency, will be evident in the metric.

Effective Habitat Time Series

Analysis of habitat time series data can be complicated because multiple baseline and alternative time series exist for different life stages of each species. An alternative may have a positive effect on one life stage but be detrimental to another. In the absence of information on habitat bottlenecks, an investigator may find it difficult to determine if changes in habitat available for a life stage will ultimately affect the population.

The effective habitat time series (EHTS; Bovee 1982) is a modified version of a habitat time series designed to help address the problem of nonuniform effects of habitat availability for different life stages or trophic levels. The EHTS is a simplified population model that is responsive to changes in habitat availability over time. The effective habitat model is based on the concept of a habitat ratio, predicated on the idea that all life stages do not need the same amount of habitat to sustain a certain standing crop of adults. A small amount of spawning habitat can produce many or just a few adults, for example, depending on survival from egg to adult life stage. If a little spawning habitat will produce many adults, relatively less spawning habitat is needed compared to adult habitat, because adults require more space than fish eggs do.

Although habitat ratios can be approximated using a bit of intuition and professional judgment, they should be based on growth, density, and survival rates for the population at hand. In the effective habitat time series, habitat ratios are usually based on average values of these population parameters and are treated as constants. More sophisticated population models relate growth and survival to densities calculated on the basis of population size and habitat area. The biological information needed for empirical estimation of habitat ratios includes:

1) The average weight of fish in each age class of the population.
2) The periodicity of the population.
3) The life span and average age of adults at maturity.
4) The average number of eggs deposited by the average spawning female (fecundity adjusted for proportion and average size of spawning females).
5) Average density of spawners (D_{sp}) per unit of spawning habitat (H_{sp}), including corrections for multiple spawners using the same area.
6) Average survival of eggs to the fry life stage (S_{egg}).
7) Average numerical or biomass density of fry (D_{fry}) per unit of fry habitat (H_{fry}). (Note: It may be advisable to estimate densities monthly for the first growing season, or subdivide the life stage into fry/fingerling.)
8) Average survival of fry to the juvenile life stage (S_{fry}). If using monthly density estimates, you should also use monthly survival.

9) Average numerical or biomass density of juveniles (D_{juv}) per unit of juvenile habitat (H_{juv}). If maturation does not occur at age 2, juvenile densities should be determined by age.
10) Average survival of juveniles to adults (S_{juv}). If maturation does not occur at age 2, survival rates should be determined by age class.
11) Average numerical or biomass density of adults (D_{adult}).
12) Average annual survival of adults (S_{adult}).

The procedure for calculating habitat ratios starts with an estimate of an average number of adult fish per unit habitat area (referred to as the adult habitat density). Adult habitat density can be averaged from measurements of adult numbers and habitat area or determined from a regression similar to the one shown in Fig. 4-25. Given a reference amount of adult habitat, say 10,000 m^2 (1 ha), the average adult habitat density is used to estimate how many adults could be supported, on average, in that much space.

Adult habitat density can also be derived from biomass estimates, provided that the investigator also has access to age and weight data. For example, suppose the estimated average adult biomass density for a stream is 650 kg/ha of available adult habitat. If we know the average weight of adult fish in each age group, we can convert the biomass density into an equivalent numerical estimate of first year adults (Bovee 1982):

$$N_I = \frac{B}{w_I + w_{II}S + w_{III}S^2 ... w_n S^{(n-1)}} \quad (35)$$

where N_I is the equivalent numerical estimate of first-year adults representing a population with a given size structure and biomass B, w_I, w_{II}, w_{III}, w_n are the average weights of each adult age group, and S is the average annual adult survival rate.

The number of juveniles needed to produce the requisite number of first-year adults is found by dividing N_I by the average annual survival rate of preadult juveniles (the last age class of immature fish before reaching adulthood):

$$N_J = \frac{N_I}{S_J} \quad (36)$$

where N_J is the number of prerecruitment juveniles needed to produce an adult year class of N_I with a juvenile survival rate, S_J. By the same logic, the number of fry required is:

$$N_F = \frac{N_J}{S_F} \quad (37)$$

where N_F is the number of fry needed to produce the requisite number of juveniles, N_J, given a fry survival rate, S_J. Note that if there are several age groups of juveniles, equation 36 is simply stepped backward for each age group until we reach the fry life stage. Furthermore, there may be several within-year habitat ratios for fry to account for rapidly changing first-year survival rates and average densities.

Eventually, following the same process, we calculate the number of eggs needed to produce the required number of fry. The number of adult spawners (i.e., females) needed is calculated by dividing the required number of eggs by the average fecundity of the population. The amount of spawning habitat (H_{spawn}) needed to provide unimpeded (e.g., no superposition of nests) spawning is calculated by dividing the number of spawners (N_{spawn}) by average spawner density (D_{spawn}):

$$H_{spawn} = \frac{N_{spawn}}{D_{spawn}} \quad (38)$$

Likewise, if we know the average habitat density for each life stage, we can calculate the amount of habitat needed for fry, fingerlings, and however many age groups of juveniles we have. Habitat requirements are computed by dividing the required number of fish by the average density for the life stage.

To calculate the habitat ratios, we have to go back to the initial value of adult habitat. For example, suppose that by the time we get to equation 38, the habitat required for spawning (H_{spawn}) was 100 m^2. This estimate was based on an initial 10,000 m^2 of adult habitat. Therefore, the ratio between spawning habitat and adult habitat is 100:1.

One way to interpret this ratio is that for every square meter of effectively used spawning habitat, you will eventually need 100 m^2 of adult habitat. Alternatively, to sustain an adult population at the carrying capacity of 10,000 ha of adult habitat, around 100 ha of spawning habitat will be required. For EHTS, habitat ratios are calculated from life stage to life stage (e.g., spawning to fry to juvenile).

An effective habitat time series (Table 5-2) is created by comparing the habitat available for a life stage with an estimate of the amount of habitat needed to support the number of animals present during each timestep.

The first step in filling out the EHTS table is to record the amount of habitat available for each time-step and life stage from the habitat time series. Annual or subannual habitat values may be represented by any of the temporal metrics from the habitat time series. The amount recorded depends on the kind of relationships assumed to exist between the habitat available during the year for a life stage and the number of animals of that group. For example, one might use the minimum effective spawning habitat that occurred during the spawning-incubation period, the 90% habitat exceedance value from May through July for fry, and the average of the four lowest consecutive months during the growing season for juveniles and adults. Typically, one habitat value is recorded for each year, although it is also possible to subdivide the year. The same habitat metrics for each life stage must be used consistently when comparing alternatives. In Table 5-2, the annual habitat availability metrics are shown in bold print.

Table 5-2. Computation procedure used to develop an effective habitat time series.

Life Stage	Year 1	2	3	4	5	6	7	8	9	10
Net Effective Adult @ t-l		650	950	700	400	400	980	1135	1250	1278
Spawning										
Available	**10**	**12**	**6**	**2**	**2**	**4**	**1**	**10**	**8**	**4**
Required (1:340)	?	1.9	2.8	2.06	1.18	1.18	2.88	3.34	3.68	3.76
Effective	*10*	*1.9*	*2.8*	2	*1.18*	*1.18*	*1*	*3.34*	*3.68*	*3.76*
Fry										
Available	**250**	**400**	**925**	**800**	**900**	**940**	**990**	**200**	**500**	**735**
Required (85:1)	850	162	237	170	100	100	85	284	312	319
Effective	*250*	*162*	*237*	*170*	*100*	*100*	*85*	*200*	*312*	*319*
Juvenile										
Available	**525**	**650**	**900**	**950**	**900**	**950**	**990**	**350**	**600**	**825**
Required (1:1)		250	162	237	170	100	100	85	200	312
Effective		*250*	*162*	*237*	*170*	*100*	*100*	*85*	*200*	*312*
Adult										
Available	**650**	**950**	**700**	**400**	**400**	**1245**	**1225**	**1775**	**1925**	**1750**
Recruitment (4:1)			1000	648	948	680	400	400	340	800
Carryover (75%)			712	525	300	300	735	850	938	958
Total required			1712	1173	1248	980	1135	1250	1278	1758
Net Effective Adult	650	950	700	400	400	980	1135	1250	1278	1750

The next step in the development of an EHTS is to compute the required habitat amount for each life stage based on the habitat ratio for the life stage and availability of habitat for the previous life stage. For example, in the first year there are 10 units of spawning habitat available. If all of the spawning habitat is used (which is the initial assumption in beginning an EHTS), there is a potential demand for 850 units of fry habitat during the same year, based on a spawning-to-fry ratio of 1:85. The effective fry habitat is the lesser of the available and required amounts (250). The effective habitat for fry in year 1 (250 units) translates into an equal area required for juveniles in year 2, owing to the 1:1 habitat ratio between fry and juvenile habitat types. The time series is staggered again when the juveniles from the second year mature in year 3. The total amount of adult habitat needed in year 3 is a combination of habitat for new recruits (equal to four times the juvenile effective habitat for the previous year) plus the adult habitat required for last year's surviving adults (calculated as 75% of the previous year's effective adult habitat). The net effective adult habitat is the lesser of the available and total required adult habitat for year 3. To calculate the amount of spawning habitat needed in year 4, the adult to spawning habitat ratio (340:1) is applied to the previous year's net effective adult habitat. In this case, it would take a little over two units of spawning habitat to produce enough surviving eggs to replace the adults that could have been supported in the available habitat during year 3.

The EHTS incorporates memory into the habitat time series by linking the habitat requirements of the different life stages in time. In Table 5-2, the paucity of spawning habitat in year 7 is reflected in a reduced habitat requirement for adult recruitment in year 9. The only reason that the overall adult habitat requirement is fairly high in year 9 is because there was a large carryover of adults from years 7 and 8. If spawning habitat had been abundant during year 7, the demand for adult habitat in year 9 would have been much larger. The incorporation of time links in the EHTS makes it more sensitive to the sequence of events, not just their magnitude. For example, this technique could be used to evaluate the consequences of several poor spawning years in a row. The EHTS also tends to smooth or buffer the effects of a single catastrophe or windfall, as illustrated in years 7 and 8. During these 2 years there was a ten-fold difference in availability of spawning habitat, but the net change in effective adult habitat was only about 50%.

Despite its potential as a decision-making aid, the EHTS model is based on several simplifying assumptions that may limit its usefulness. Habitat ratios are based on average

weights, growth rates, age structures, densities, fecundities, and survival rates. All of these variables are treated as constants. Compensatory and density-dependent growth, reproduction, and survival functions are impossible to incorporate into such a simple model. The EHTS model does not distinguish between acute and chronic habitat bottleneck effects. These influences can be explicitly incorporated through the selection of an appropriate habitat time series metric (e.g., average annual habitat value, annual minimum, average of lowest 50% of values), but the use of a single annual value for each time-step essentially precludes the evaluation of within-year cumulative effects. Judgments regarding the chronic or acute influences of habitat restrictions are the explicit responsibility of the investigators using the model. Most vexing is the fact that data for the calculation of habitat ratios for all life stages may be difficult or impossible to obtain.

Waddle (1992) used a cohort table (e.g., Table 5-1) and a habitat time series for concurrent time periods to calculate average numerical densities, sizes, and survival rates of a population. From the habitat ratios derived from these averages, the EHTS and the adult trout population were plotted on the same graph. In theory, the EHTS is a surrogate for the population, so if the habitat ratios are approximately correct, the two time traces should be synchronous and auto-correlated. That is, when the EHTS goes up, the adult population should go up in the same time-step and vice versa. If there are conspicuous lags between EHTS and adult population, the emphasis was likely placed on the wrong life stage via the habitat ratio or the time to maturation was misjudged, resulting in too many or too few age groups between egg and adult. Access to a cohort table allows the investigator to adjust these variables until a reasonable agreement is found between EHTS and population variations. This is basically the same process used to calibrate more sophisticated population models (Cheslak and Jacobson 1990; Williamson et al. 1993).

Once the EHTS model is calibrated, the output can be used to quantify differences between baseline conditions and an alternative following the same procedures described for a simple habitat time series analysis. Because the EHTS embodies year-to-year variation explicitly, however, the analyst should use all of the values in the EHTS to calculate impacts (i.e., use of trimmed means or the average of the lowest 50% of the values would be meaningless in this type of analysis).

Feasibility and Risk Analysis

The analysis of feasibility and risk are combined in this section because the two subjects are almost inseparable. There are two basic approaches to risk planning: overdesign and risk containment. Overdesign is more appropriately termed risk avoidance, because the goal is to reduce the probability of failure to as near zero as possible. We appreciate overdesign whenever we get into an elevator, but this is not a very useful approach for instream flow studies. The most serious drawback of overdesign is that it can undermine the economic feasibility of an alternative. The second major disadvantage of overdesign is that there is usually no strategy for what to do when the "fail-safe" alternative fails. The failure then becomes a crisis and solutions are cobbled together in the eleventh hour.

Risk containment operates under the assumption that all alternatives will fail sooner or later, and it is better to plan for failure than to hope against it. Consequently, a contingency plan is incorporated as an integral part of the alternative. During negotiations, the concept of risk containment can be a powerful tool because low-risk alternatives tend not to be very effective for providing habitat. For example, under the concept of risk avoidance, someone could recommend an alternative that produces no habitat but could be guaranteed every year. Risk containment is a more flexible solution because failures are anticipated. And because failure is planned into the alternative, many potential crises can be averted.

The approach taken to analyze feasibility and risk in IFIM alternatives depends somewhat on whether the change being evaluated is time-independent. That is, the management for one month is not physically affected or constrained by alternatives implemented during a previous month. This type of analysis is exemplified by an analysis of the impacts associated with a constant diversion from an unregulated stream. There is no opportunity to shift around the storage and release of water, and the feasibility of different alternatives will be dictated primarily by the unregulated flow regime. Alternatives that involve reservoir operations or institutionalized adjudications of water rights are examples of analyses that are time-dependent. That is, management decisions made during one time-step will constrain, and may preclude, feasible options during the next time-step.

Time-Independent Risk Assessment

The flow duration curve is the primary tool used to analyze feasibility and risk of alternatives in time-step independent situations. Often, this kind of assessment is conducted to evaluate the risk of violating a state's instream flow standard or a recommended minimum flow for a stream. Rather than evaluating conditions for the entire year, the assessment concentrates on the time of the year when the risk is highest. For example, consider the time of year when an agricultural diversion in an unregulated stream might have its greatest impact. In most parts of the United States, July and August are prime candidates because streamflow is often on the wane, while the demand for irrigation water is still high. In this case, we separate flow duration curves for July and August to test the feasibility of an instream flow recommendation or to determine the risk that the diversion would violate a State instream flow standard. The feasibility analysis could be designed to evaluate each

month separately, or both months could be considered as a single time period if low flows of the same magnitude occurrred in both months.

A time-independent risk assessment is illustrated in Fig. 5-5. In this example, an irrigation district proposes to build a diversion canal to deliver a 95% firm yield of 28 million m^3 of water to a new tract of farmland. The projected demand in August is 5.5 m^3, or an average daily flow of about 2.1 cms. An analysis of habitat impacts associated with the project was conducted by State and Federal fish and wildlife agencies as part of the permitting process under Section 404 of the Clean Water Act. On the basis of this analysis, the resource agencies recommended an instream flow of 2.8 cms during August. What is the probability that the natural inflow will be insufficient to meet both demands during August?

The combined demand for the diversion and the instream flow standard amounts to a little less than 5 cms. The good news, according to Fig. 5-5, is that the natural August inflow for this river exceeds 5 cms about 75% of the time. This means that in 3 out of 4 years, on average, there will be no shortage and no conflict between the diversion and the instream flow requirements. The bad news is that the alternative will not be feasible in about 1 out of 4 years. This is where a contingency plan might be appropriate.

Assuming that a 25% failure rate is not acceptable, one of many possible contingencies could be to redefine the firm yield for August to 4.4 m^3 and to relax the instream flow requirement to 1.4 cms whenever the natural inflow was less than 5 cms. With this contingency plan in place, the combined demand could be met in 9 out of 10 years (Fig. 5-6). Variations on the proposed alternative and contingency plan might include allowing larger diversions during July when water is more plentiful, building off-channel storage facilities, growing crops that mature earlier in the season, or developing another contingency plan for a 1-in-10 year drought.

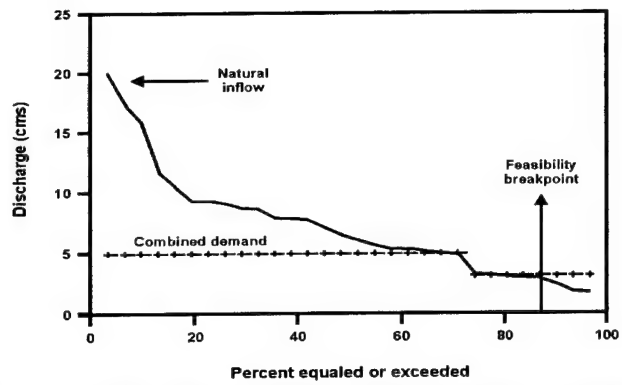

Fig. 5-6. Flow duration curves for August natural inflow and combined instream flow and diversion demands, used to test feasibility of a two-staged contingency plan in an unregulated stream. This type of analysis is typical of a time-independent risk assessment.

Time-Dependent Risk Assessment

The most common form of time-dependent assessment involves the operation of one or more reservoirs in a hydrologic network. In order to decide when to store and release water, reservoir managers rely on rule curves and reservoir operations models. Rule curves consist of sequences of storage goals to be attained at various times of the year for different water use objectives. Rule curves are typically designed to incorporate the needs of all recognized water users, often on a prioritized basis. Reservoir operations are normally governed by several rule curves, which are dependent on anticipated water supplies and demands (Fig. 5-7). During a dry year, for example, storage is maximized in order to guarantee the availability of water

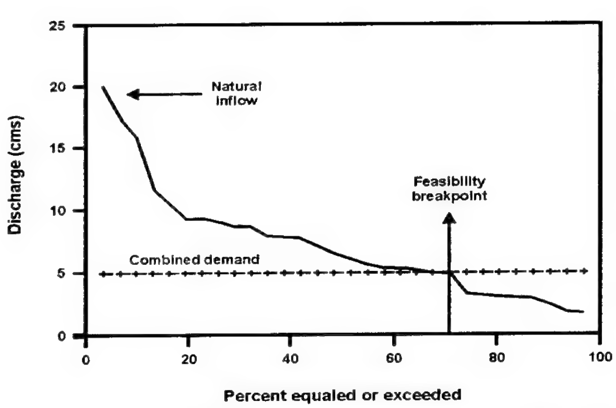

Fig. 5-5. Flow duration curves for August natural inflow and combined instream flow and diversion demands, used to test feasibility of a proposed alternative in an unregulated stream. This type of analysis is typical of a time-independent risk assessment.

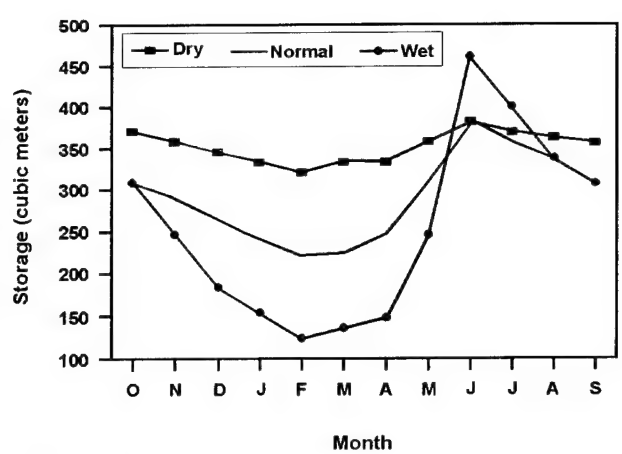

Fig. 5-7. Rule curves for reservoir operations during below-normal (dry), normal, and above-normal (wet) periods of water supply.

when it is needed. In contrast, storage is reduced during wet years to capture high runoff events, thereby satisfying two demands at once (flood attenuation and active storage demands). In addition to rule curves, there must also be operational rules or guidelines to tell the operator which rule curve to follow at any particular time. These guidelines, called trigger levels, are based on several factors, but the most important include prior conditions and forecasting capabilities.

A reservoir operations (RESOP) model is usually the most flexible and reliable method for testing reservoir operating rules. Regardless of their sophistication, these RESOP models are all based on the concept of a water budget (Fig. 5-8). The reservoir mass balance can be summarized as:

$$S_{(t+1)} = S_t + Q_{in} - Q_{out} \pm E \qquad \textbf{(39)}$$

where $S_{(t+1)}$ is storage target from the rule curve for time $(t+1)$, S_t is the current volume in storage at time (t), Q_{in} and Q_{out} are the inflow and outflow volumes, respectively, that occur in the time interval from t to t+1, and E is the effective evaporation during the time interval. Note that effective evaporation is the difference between precipitation and actual evaporation during the time-step and may be positive or negative.

Reservoir outflow for a time-step is determined by obtaining a storage goal from the rule curve and estimating the inflow and effective evaporation for the time period. In using the reservoir operation model to establish operating rules, historical inflows and effective evaporation rates are used as inputs, and outflow demands are varied experimentally to determine the resultant storage volume. An iterative procedure is used to determine the periodic storage levels (usually monthly) that most reliably meet reservoir demands, given a particular pattern of inflow and effective evaporation. In some of the more sophisticated RESOP models, the inflow is manipulated to develop "worst case" rules curves to handle extreme conditions, such as several drought years in sequence.

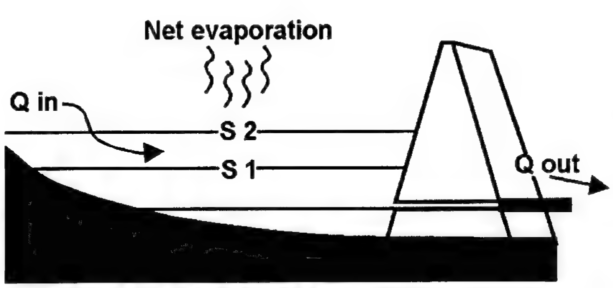

Fig. 5-8. Components of a reservoir mass balance model. S1 represents the volume in the reservoir at time (t) and S2 is the storage goal from the rule curve at time (t + 1). Q in is inflow and Q out is outflow.

In the process of experimenting with different inflow and demand scenarios, patterns often emerge that assist in forecasting when the operating rules should be triggered from one rule curve to another. The ability to forecast water supplies is a major determinant of operational flexibility. In hydrologic systems dominated by snowmelt, the amount of inflow can be determined with considerable accuracy, well in advance of runoff. For such systems, it is relatively easy to anticipate whether the wet, normal, or dry year rule curve should be followed. In contrast, inflow is uncertain in systems driven by thunderstorms. It may not rain for 3 months and then it could rain 10 cm overnight. Reservoirs located in precipitation regimes such as this are often operated on the basis of current storage and predicted rainfall. Unless the reservoir is very large compared to potential inflow, decisions on storage targets are often made and revised daily.

Two valuable pieces of information are gained from the RESOP model. First, the investigators can observe the behavior of the reservoir mass balance under a variety of inflow patterns, water demands, and operating rules. The goal of these simulations is to keep reservoir storage within specified operating boundaries as much of the time as possible: operating rules that result in frequent releases over the spillway or that draw reservoir storage down below the lower operating limit are modified or rejected. Second, flow duration curves of the outflow can be generated for different operating rules and inflows. From the flow duration curve, one can determine the deliverable firm yield. Through repetitive simulations, it is possible to determine rules which provide the most water with the greatest certainty.

Contingency Plans

The concept of contingency planning was illustrated in Fig. 5-6 for a simple case of a two-staged instream flow recommendation. At the risk of redundancy, a contingency plan can be defined as an alternate alternative, essentially an "if-then" statement. Several layers of contingencies may be designed to handle different situations, so long as the layers do not interfere with decision-making. For example, one contingency might be designed for a 1-in-5 year drought, another for a 1-in-10 year drought. Although layering contingency plans is a good idea, there is a point of diminishing returns. At some point, the probability of failure becomes so small that it may no longer be worthy of debate or planning. The probability level at which this happens is determined in part by value of the resource and in part by the tenacity of the negotiators.

Once a contingency plan has been identified and tested, it is important to define the conditions that trigger a move from the basic alternative to the contingency plan or from one contingency plan to another. Reservoir storage is often used as a criterion for triggering a contingency plan.

Sometimes, alternative operating conditions are defined by snowpack or antecedent rainfall. A drought could be signaled if the system has received little precipitation during a normally wet period. In such a circumstance, it might be wise to move to the drought contingency before reservoirs are severely drawn down or streams dry up. Conversely, if snowpack is three times normal levels, it would be senseless (and probably illegal) to maintain reservoirs at full pool levels. Any number of criteria could be used to trigger a contingency. The important concept is that the criteria are measurable and monitored. You should know how much snowpack or rainfall the watershed normally receives and how much variation constitutes a normal, wet, or dry cycle. You should negotiate and agree on when critical decisions and measurements are to be made, and finally, what condition or combination of conditions will trigger the contingency plan.

The second important rule about contingency plans is to define the conditions that initiate movement from a contingency plan back to the original alternative. Often, the same criteria are used to move to and from the original alternative, but there is no requirement that they must be the same. It is important to define this rule because the original alternative may be conveniently forgotten once a contingency has been triggered.

The third, and perhaps most important rule, is to document your decisions throughout the process of formulating, testing, and negotiating alternatives. When people really get into the spirit of iterative problem solving, there is a tendency for alternatives, contingencies, and implementation rules to multiply like gerbils. If you do not document decisions and agreements, you may discover the phenomenon of selective memory. People may tend to remember and promote the alternatives that were most favorable to their preferred objectives.

Networks

The primary features of network habitat problems that distinguish them from single-segment analyses include cumulative impacts, synergism, and feedback mechanisms. Cumulative impacts originate from dispersed sources, the effects of which are usually additive on riverine resources. Synergism occurs when two or more projects produce an effect that neither could have produced by itself. Feedback occurs when management options in one part of the system are contingent on operations in another part, such as the operation of multiple reservoirs in a drainage basin. Even if the reservoirs are arranged serially, the contingencies, and their feedback mechanisms, must be analyzed using models developed to handle the water supply, routing, and storage network. In such situations, the effects on habitat of combined reservoir operations can be directly shown using network models.

A basinwide perspective of the biological component is also necessary to answer new questions relating to biological connectivity or continuity. These issues involve matching requisite habitat components for each life stage through time and space throughout the basin. Issues related to movement and continuity make it important to monitor the distribution of habitat in the network, not simply the amount. Complex problems of tradeoff analyses arise: Can habitat for a species be sacrificed in one geographic area to preserve or improve habitat in another area? What is the significance to the population if juvenile fish habitat is abundant in some areas of the basin and scarce elsewhere? From the larger perspective of alternatives analysis, we still evaluate alternatives on the basis of effectiveness, feasibility, and risk, whether at the segment or network scale. However, we must take into account the effects of feedback, synergism, and cumulative impacts at the network scale, whereas we can often conveniently ignore them at the segment scale.

Components of Network Habitat Analysis

Network habitat analysis has eight components, many of which are common to a segment analysis (Fig. 5-9). The driving force of the network habitat analysis is the network flow model. The flow model accounts for all water moving throughout the network in both space and time, schedules reservoir operations, accounts for all forms of water rights and delivery requirements, and records when and where violations to the rules are occurring. The flow model directly regulates both the microhabitat model and the macrohabitat model. Steady-state reservoir operations and mass routing models are used to handle the flow component of the network habitat analysis. They are typically not "first-principle" models, which simulate watershed runoff events, evapotranspiration, and groundwater recharge,

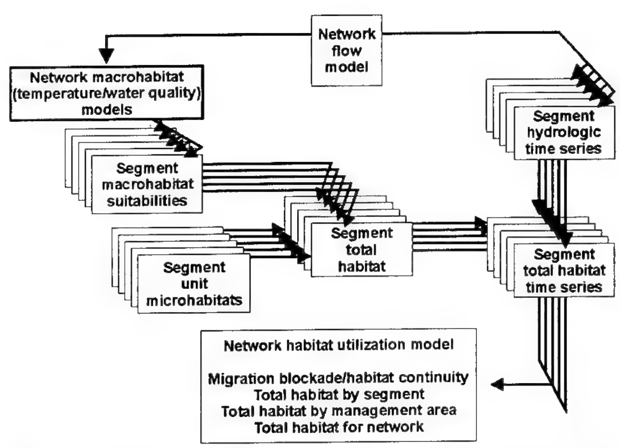

Fig. 5-9. Components of a network habitat analysis. Each layer in the figure refers to the development of a complete habitat time series model for each segment in the network. The network habitat utilization component represents the analysis of habitat continuity to evaluate issues of habitat availability versus usability.

though such detailed models do exist. The water-routing aspect of these models is conceptually simple in that they operate on an instantaneous bookkeeping method that monitors total volume of water moving through the network.

Water systems management models have been used for many years by the U.S. Army Corps of Engineers and the U.S. Bureau of Reclamation to facilitate systems design and day-to-day operations of an integrated set of multipurpose reservoirs. Originally designed for narrowly focused purposes, such as flood control, their capabilities have grown to handle power generation, industrial and municipal demands, recreation, irrigation, and other water uses, such as aquatic habitat preservation. Their logic is to meet clearly defined objectives at specified locations for water storage and deliveries from storage over a period of years. It is this multiyear focus that makes these models complicated.

The models simulate the operation of the system under different water supply and management scenarios. Some models allow the path taken by the water to be externally defined by user-supplied data, while others are "hardwired." That is, the programs can only be applied to a particular streamflow network or basin. The output from these models generally consists of tables organized by node and time. These tables usually contain predicted flows in the streams at each node, the amounts of water stored in the reservoirs, and summaries of the success or failure of meeting the delivery requirements or other operating goals. This information is carried to other components of a network habitat analysis.

The water temperature and water quality models (SNTEMP, QUAL-2E) are network models in their original configurations. In fact, single-segment treatments of temperature and water quality variables are the exception rather than the rule. When a proposed action will change the flow regime somewhere in the segment, the hydrologic change is first propagated through the network flow model. The flow for each time-step and segment is then transmitted to the macrohabitat network model. What the user normally sees as output from the network model will be discharges, temperatures, and water quality variables arrayed by time-step for each segment. It is worth remembering that these variables originated from a network simulation, because all of the changes you see at the segment level embody the feedback mechanisms occurring throughout the network. The output can be deceptively oversimplified in that what seem to be linear changes to the system can have distinctly nonlinear effects.

Total habitat time series are determined essentially the same way in a network analysis as they are at the segment level. Recall that the total habitat for a life stage in each segment is calculated according to equation 32 or 34 (Phase III). The hydrologic time series (baseline and alternatives) for the segment are obtained from the network flow model, and a habitat time series for the segment is constructed using the same techniques described in Fig. 5-1.

The simplest form of network habitat analysis is the development of a single time series that accounts for the network habitat available for the target species. This time series is developed by adding the total habitat values for all of the segments for corresponding time-steps (Table 5-3). Once the network data are in habitat time series format, they may be analyzed using the same duration statistics described previously, provided the species can complete its entire life cycle within the boundaries of the network.

Habitat Connectivity

If one part of the life cycle is isolated within one part of the network, total habitat within the network is contingent upon biological connectivity, or the accessibility of all parts of the network to all life stages. In Fig. 5-9, this step is referred to as network habitat utilization.

There are several potential connectivity problems in a network, but two are most common. The first is a flow-related passage barrier at some location within the network (Fig. 5-10). This type of barrier prevents migration at very low or very high discharges, and it usually affects fish that migrate upstream to spawn. At very low flows, the water becomes too shallow for the fish to cross over the barrier and at high flows, it gets too fast. Passage restrictions can be analyzed using the hydraulic simulation models incorporated in PHABSIM (provided that a transect was placed across the potential passage barrier). Neither PHABSIM nor IFIM, however, keeps track of the linkage between what is going on at the critical passage barrier and what is happening in the rest of the segment. Therefore, the investigator is responsible for monitoring the accessibility of different parts of the network under each alternative examined. For example, suppose that most of the spawning habitat in the network shown in Fig. 5-10 was located upstream from the passage barrier. Furthermore, assume that the passage barrier becomes impassable to the target species at flows

Table 5-3. Summation of total habitat time series data across segments to calculate a network total habitat time series.

	Total habitat				
Time step	Segment 1	Segment 2	Segment 3		Gross network
1	6,236	+ 15,235	+ 20,150	=	41,621
2	4,334	+ 12,344	+ 19,235	=	35,913
3	3,575	+ 10,932	+ 17,985	=	32,492
4	3,213	+ 11,815	+ 16,544	=	31,572
5	3,390	+ 13,440	+ 17,635	=	34,465
6	3,780	+ 12,210	+ 15,222	=	31,212
7	4,590	+ 9,835	+ 14,940	=	29,365
8	7,313	+ 8,718	+ 12,135	=	28,166

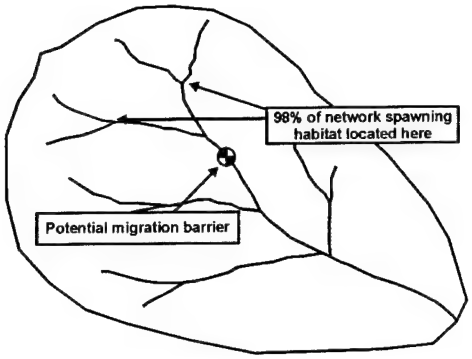

Fig. 5-10. Example of a habitat connectivity problem in a network resulting from insufficient streamflow to permit passage to spawning areas. In this case, there is a potential migration barrier in the middle of the network that becomes impassable if the flow is less than 2 cms.

less than 2 cms. When we are calculating the accessible network habitat, we count spawning habitat that occurs upstream from the passage barrier for all discharges greater than 2 cms. For flows less than 2 cms, the spawning habitat upstream from the barrier is zero, no matter how much spawning habitat is actually up there. If the fish cannot get to it, it cannot be counted as habitat for them.

Another version of a potential passage block is illustrated in Figure 5-11. Most of the spawning and fry rearing habitat is located in four headwaters streams, but the bulk of the adult habitat occurs in the mainstem and larger downstream tributaries. A new reservoir is proposed in the middle of the network. There are several habitat connectivity questions to ask in this case. First, can enough adults migrate to the spawning grounds to sustain the species at current population levels? Part of the parent stock will originate from the adult habitat that remains above the new reservoir, but will this be enough to sustain the population if no adults migrate from below the new reservoir? Second, will young fish be able to migrate downstream as they mature to recruit into areas of adult habitat below the new reservoir? Third, if the new reservoir completely isolates the headwaters population as a closed biological network, what are the implications with respect to genetic mixing? (IFIM cannot address the third question, but it ought to be asked anyway.)

One of the most interesting aspects of habitat continuity occurs with species like the striped bass (*Morone saxatilis*). The eggs from the striped bass are buoyant and incubate as they drift downstream. Hatching occurs in about 2 days at temperatures 18°-19° C and in about 3 days at temperatures in the 14°-16° C range (Scott and Crossman 1973). Continuous suspension of striped bass eggs appears to be essential for successful hatching. Without sufficient current, eggs settle to the bottom and suffocate for lack of dissolved oxygen or from siltation (May and Fuller 1965). Mortality can also occur, however, when the fry hatch in areas where fry habitat is limited or absent (Crance 1984).

An example of this type of continuity problem is illustrated in Fig. 5-12. In this case, the thermal regime is the only difference between two alternatives. Network hydrology is unchanged, so the travel time through the system is the same under both scenarios. Temperatures are slightly warmer under alternative 1, causing the eggs to hatch in 2 days. Under alternative 2, temperatures are slightly cooler and hatching occurs in 3 days. The travel time from the spawning grounds to the area of fry rearing habitat is 3 days under the prevailing flow regime for this time period. Higher mortality would be expected under alternative 1, because newly hatched fry are still a day away from their rearing habitat. Under alternative 2, the eggs arrive at the rearing habitat just about the same time that they begin hatching.

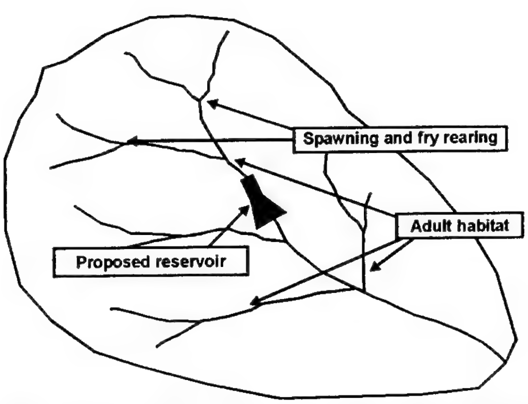

Fig. 5-11. Potential isolation of a small, closed biological network resulting from the construction of a new reservoir.

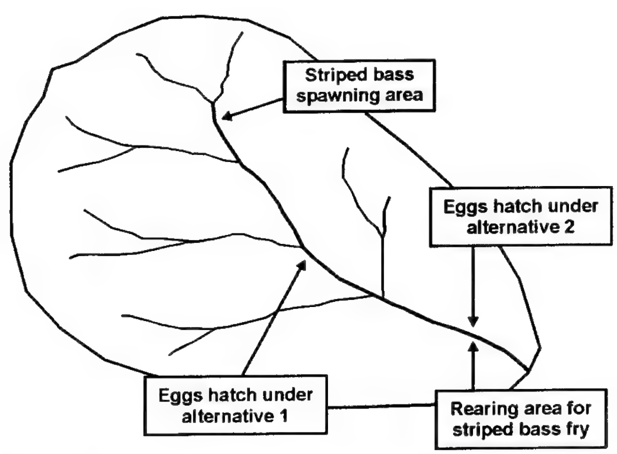

Fig. 5-12. Habitat connectivity relationships between locations of spawning and fry-rearing habitats for striped bass, temperature regimes, and network travel time.

Network utilized or accessible habitat, as we have defined it, should be distinguished from network available habitat whenever issues of habitat connectivity are apparent. Although there is no computer model to help you determine habitat accessibility, such a determination is still required in the calculation of total usable habitat. We have seen why the simple summation of available habitat is not always sufficient to describe total usable habitat. When you are in the process of determining availability and accessibility, you should consider the following issues related to biological connectivity:

1) Species phenology
   - important life stages and potential bottlenecks (e.g., r-limited or k-limited?)
   - temporal and spatial distribution of life stages
   - life stage interdependencies (i.e., must have successful spawning to have egg incubation)
2) Spatial connectivity
   - areas of dewatering or high flows (velocity barriers)
   - structural blockages (dams, weirs, waterfalls)
   - macrohabitat blockages (temperature, DO)
3) Temporal connectivity
   - temporal accumulation of life requisites (temperature, degree-days, extended growth conditions)
   - timing of certain threshold temperatures (e.g., water temperatures must reach X degrees before spawning)
   - travel times, settling velocities, and temperature relations for semibuoyant eggs or other forms of biological flotsam

Negotiating Strategies

Three basic negotiating strategies are commonly employed during natural resource bargaining sessions: competitive, cooperative, and integrative. It is important to quickly identify the strategy being pursued by your counterparts at the bargaining table. If you are engaged in cooperative or integrative negotiating and your opposition is operating competitively, you risk being trampled (at least figuratively).

The competitive strategy is marked by extreme positional bargaining. Personal values are often more important to individual stakeholders than scientific fact or reason. Concessions are small and hard-fought, if they are given at all. From an instream flow perspective, you will be able to identify a competitive negotiation when you hear expressions such as "this is a waste of our (my) water," the keywords of a position based on personal values being "waste" and "my water." If you are involved in an extremely competitive negotiation, you might want to seriously consider your BATNA. You might decide that an arbitrated solution is far more attractive than the hassle that usually goes along with a competitive negotiation. Arbitration is not always a way out of a competitive negotiation, however, because most arbitrators will encourage contending parties to come to an agreement on their own.

Cooperative negotiations are much friendlier than the competitive type, being identifiable by the existence of a *quid pro quo* attitude. The basic idea of a cooperative negotiation is reciprocity: give something so that you receive something in return. However, even cooperative negotiations have elements of competition. People still keep score, and although the atmosphere is less confrontational, there is also an underlying desire to get more than you had to give.

Positional bargaining may be evident in cooperative negotiations, but positions will generally be softer than they are in competitive situations. You are more likely to hear the preamble to a position stated something like, "Our agency's policy is to . . . " or "Our clients expect us to . . . " Positions are more likely to be depersonalized in a cooperative negotiation.

Gamesmanship is more apparent in cooperative bargaining than in either of the other types. Typically, the game consists of building throw-away positions and concessions. The idea is to develop a concession that means absolutely nothing to your interests, bargain mightily over it, and finally give in to your counterpart across the table. The reason for doing this, of course, is to serve the moral obligation of reciprocity back to your opponents. Because you have conceded on a point that is "crucial" to your interests, the socially acceptable behavior on the part of your opponents would be to offer a concession in return. The wise negotiator will recognize, of course, that the opponents are doing exactly the same thing.

Integrative negotiations are the most difficult to achieve but are generally the least stressful on participants. The identifying characteristic of the integrative negotiation is a lack of positional bargaining. Participants are encouraged to develop joint ownership of all of the issues, goals, and values embodied in the IFIM problem. For example, the irrigation district proposing a new water supply reservoir would be as concerned about fisheries issues as the state wildlife agency in a truly integrative negotiation. Conversely, the wildlife agency would share the same concerns about agricultural production. The key to integrative negotiation is the ability of *all* parties to lay aside their own personal and agency values for the good of the negotiating team and the pursuit of the elegant solution. Integrative negotiating teams not only develop idealized objectives, they actually believe in them. Unfortunately, integrative negotiations are rare because they depend on extremely strong trust relations among all of the participants. At the first hint of positional bargaining, the negotiation will quickly degrade back to (at least) a cooperative strategy.

Negotiating Tactics

Some negotiating tactics are used to help the stakeholders advance to an agreement. Others are used in an attempt to obtain an advantage over one's opponents, especially in competitive and cooperative negotiations. Special

advantages are unnecessary in integrative negotiations, so you will rarely if ever see any negative tactics employed in this setting. Once you know what to look for, you should be equipped to deal with competitive negotiation tactics when you encounter them.

The salami technique is one of those tactics designed to help negotiators achieve an agreement. This benign, usually helpful tactic does not impart advantage or leverage to one party or another. Under this approach, the problem is dissected into manageable slices and the easiest problems are resolved first. The theory of the salami technique is that nothing promotes success so much as success itself. Stakeholders will have made an investment in the process and, in theory, will be more willing to attack more difficult problems.

Brainstorming is a tactic most commonly used in integrative negotiations, but it may also appear in cooperative settings. Creativity and innovative solutions are sought among members of the negotiating team during short, intense brainstorming sessions. A professional facilitator or mediator will be very helpful in recording ideas, drawing in the reluctant participants, and generally refereeing the proceedings. The purpose of a brainstorming session is to come up with as many ideas as possible to solve a problem, but this activity may have the added benefit of loosening up the participants. Because this tactic involves creativity toward solving a problem, you will almost never see it in a competitive negotiation.

Quid pro quo translates from Latin to "what for what" and refers to the practice of give and take in a negotiation. This tactic is a central feature of the cooperative negotiation strategy, which we reintroduce here to alert you to a common mistake of scientists who are new to the instream flow game. The mistake has its origins in the standard-setting arena, where the goal of the biologist is to determine the minimum flow for a segment or stream. To the biologist, minimum means minimum. It is tantamount to a "bottom line." To other stakeholders in a cooperative negotiation, however, the flow recommended by the biologist is interpreted as an opening gambit. Everyone involved in the negotiation, with the exception of the biologist, expects the negotiated instream flow to be lower than the initial offering. Furthermore, even if the biologist states explicitly that the recommendation is the absolute minimum, bottom line flow acceptable to his or her agency, no one at the table believes it. When the biologist refuses to budge from the first offering, he or she will appear obdurate and will be accused of not bargaining in good faith. This is one of the reasons that we caution people about the use of standard-setting methods when dealing with incremental problems.

We have spoken previously about the necessity of deadlines to ensure progress in an IFIM study. However, skillful negotiators will sometimes conjure up deadlines to get things moving or to keep them going. Used in this manner, artificial deadlines can actually be beneficial to a negotiation. Some types of artificial deadlines, however, can work against you. The most common artificial deadline is one you impose on yourself. Suppose you have traveled a considerable distance to attend the bargaining session. You let it slip that you must be home by Friday night because it is your wedding anniversary. If the negotiation is at all competitive, do not be surprised if your opposition stalls for most of the week. Serious negotiating will begin when you are about to leave for the airport. The way around this tactic is to leave your departure open-ended.

Several negotiating tactics are more commonly associated with competitive negotiations than with the other two strategies. These include false legitimacy, false proxy, apparent withdrawal, and good cop/bad cop. We do not endorse any of these tactics because they are designed to give one group an advantage over others in a negotiation. We present the tactics here so that you can recognize them should you encounter one or more of them during a negotiation.

False legitimacy is a tactic used to make an alternative appear to be infeasible due to factors beyond the control of the negotiator. The way that false legitimacy might look in a negotiation is illustrated in the following example. Suppose that you are negotiating the release of a channel maintenance flow from a reservoir that has not yet been built. During one of the bargaining sessions, an engineer representing the applicant states that the recommended channel maintenance flow is impossible because the outlets to the new dam are too small. To prove his point, he spreads a blueprint of the proposed dam across the negotiating table and goes to considerable lengths to explain how much water could be released from each gate. You should recognize immediately that a blueprint is not the same thing as a fully-hardened concrete dam. You can change the size of the outlet on a drawing with an eraser and pencil. Because the outlet size is specified on the blueprint, however, it looks like a legitimate physical restriction.

False proxy, apparent withdrawal, and good cop/bad cop are all tactics designed to help a negotiator learn where the opposition is willing to compromise without committing themselves to reciprocation. These tactics usually employ a team consisting of a superior and a subordinate, or in the case of good cop/bad cop, a nice person and one who is not so nice. The general approach is to encourage a spirit of empathy toward the subordinate, and then to find a reason for the superior to be absent from the negotiation. (With the false proxy technique, the superior person has "another commitment," and cannot attend the meeting. With apparent withdrawal, the superior will feign abandoning the negotiation out of frustration, leaving the subordinate behind.) The job of the subordinate is to find out where other parties might compromise, as a means of luring the superior back into the negotiation. The subordinate may even offer concessions in the spirit of a cooperative

strategy. The disingenuous feature of these tactics is that the subordinate does not have the authority to make concessions or agreements. Thus, your opponents can learn where you are open to compromise, without providing reciprocal information to your side. The best defense against these tactics is to avoid negotiation with people who do not have the authority to make binding agreements.

Negotiating for Success with IFIM

We often raise eyebrows when we state that IFIM is scientific, but it is not science. The questions raised during an application of IFIM are scientific but cannot be answered without moving beyond science into values. In addition, for a technology to be policy-relevant, it must be accepted by all sides in a dispute. Acceptance of scientific knowledge or technology is essentially a political problem. When the results of any course of action are to some degree uncertain, choice must be based on belief. Choosing a technical solution is often a matter of convincing other parties through negotiation. In addition to these basic findings, we have discovered some practices that may help you achieve success in natural resource negotiations.

- Maintain contacts on all sides and work to establish goodwill. In successful negotiations, the applicants made explicit efforts to involve all the relevant parties, maintain a dialogue with all sides, and establish an atmosphere of goodwill. This was accomplished by open communications, numerous meetings, site visits, and small agreements. Although the negotiations lagged at times, the applicants endeavored to ensure that the parties could interact and established an atmosphere in which everyone felt progress was being made.
- Be ready with understandable, scientifically based alternatives. To be successful in using tools like IFIM, the analysis must be more than merely accurate. The parties must also believe the technology is a legitimate product of science. In the Terror Lake case, some technologies were accepted by the parties as legitimate and some were not. In particular, the parties believed IFIM to be appropriate for the problem they faced and the results of IFIM were understood. In other case studies, IFIM was less influential because not all the parties believed the technique was appropriate. Acceptability of IFIM does not appear to be related to peer review, but rather to the persuasiveness of involved scientists.
- Have a capacity for institutional analysis. Scientists can use analysis to manipulate the policy process. The way an analyst states scientific questions and potential answers helps shape the policy problem. Because policy-makers are looking for solutions, one of the contributions of scientists is to array a range of possible alternatives, suggesting which is most feasible (and being prepared with a risk assessment and appropriate contingency plans). In the Terror Lake case, some scientists were able to help shape the policy problem by understanding the constraints faced by decision-makers and suggesting options that fit within those constraints. Avoid the error of undertaking studies for their own sake. IFIM will not help resolve issues if the stakeholders have not helped shape the policy problem.
- Analyze the roles and behaviors likely to be played by the parties. In the most successful negotiations we have studied, representatives of the license applicant attempted to understand not only the formal process but the informal relations among the parties. The least successful negotiations were characterized by inattention to this detail. In some instances, the negotiation was one among many and participation by the parties was often formalistic and cursory.
- Understand the rules governing the negotiation. The rules of natural resource management make all the difference. Whether the rules focus on formal meetings or encourage informal meetings, for example, makes a difference in the ease of negotiation and (probably) the outcome. The chances for resolving conflict increase where the official rules are understood, clear and consistent signals are provided by the regulator, and the parties develop an understanding of process. Changing the rules in the middle of the consultation will have a negative influence on the process.
- Encourage participation of the ultimate decision-maker. In successful applications of IFIM, there was a strong shared desire for the decision-maker to mediate, participate, or at least clearly interpret the rules of engagement. In those cases where negotiations failed, the decision-maker refused to take these steps almost without exception. Parties to these disputes felt they would have been more successful in working out solutions if the decision-maker had more actively guided the negotiations.
- Beware of outside influence and false expectations. If even a few parties in your negotiation are also involved in some other negotiation, their experiences elsewhere can raise or dash expectations. Failure in one negotiation can spill over into others that are similar in substance or participants.
- The problem must be tractable to scientific solutions. Two conditions are necessary if you plan to use a technology to aid decision-making: (1) the technology must be policy-relevant and (2) the problem must be amenable to scientific analysis. To be policy-relevant, a technology must provide information in a way useful to decision-makers. In the Terror Lake case, IFIM provided fish habitat information in a form that could be translated to water supply. The instream flow

issue was subjectable to scientific analysis because the parties wanted information on the relation between flow and fish habitat.

Because negotiation involves questions about the distribution of benefits—not only who gets which piece, but what the shape and size of the pie should be (Mnookin 1993)—the scientist is thrown into the realm of ethics. In this realm, the issues call for more than a recitation of facts or scientific conclusions. They call for prescriptions about what should be done.

The decision to behave as a scientist, manager, or policy-maker has instrumental and substantive components. Instrumental factors concern moving the negotiation forward. They include such factors as timing, deadlines, commitments, context, and ripeness. Timing considers the purely quantitative problems of when, what, and how to conduct studies. Analyses may have to be completed by certain deadlines to move the process forward. Negotiations concern commitments which may require work as a technician or policy-maker. Diagnosing the context of the negotiation is policy work. A negotiation calling for a high level of technical analysis requires focusing on scientific work. Ripeness refers to how far along you are in the negotiation. The early stages of negotiation may require strategic planning, the middle stages analysis, and the final stages balancing competing values. Judging the ripeness of a negotiation helps guide emphasis on the appropriate behavior.

Substantive components include personal skills and knowledge, personal level of authority, agency role, and solution-building. You bring to the negotiation a host of personal skills and knowledge, some from training and some from experience. If your skills and knowledge run more toward quantitative analysis you will be most useful as a scientist or technician. Although science may be your personal focus, the context of your personal authority may thrust you into the realm of managers or policy-makers. What is your personal level of authority? Are you a major player in the negotiation or a minor functionary? If you are charged with keeping the consultation on track your requirements as a policy analyst are increased.

You must also consider your agency role. Whom do you represent? What is that agency's traditional role in this issue? If your organization is expected to provide quantitative analysis you will have to do so. At the same time, you may have strategic responsibilities because of context, ripeness, or role. Solution-building is more than just formulating an effective and feasible alternative; it includes deciding which solution is best (or at least most acceptable to everyone). The most effective negotiators we found were those who understood technical issues and quantitative analysis but who were not put off by policy concerns and value differences. Integrating the two strategies is a matter of personal skill and experience and is essential for successful negotiating of natural resources issues.

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Glossary

Acre-foot - The volume of water required to inundate 1 acre of land to a depth of 1 foot, equal to 43,560 ft^3 or 1,233.5 m^3.

Active storage - That portion of reservoir storage allocated for release to satisfy a demand (e.g., irrigation, power generation, instream flows) and not reserved for flood detention or sediment accumulation.

Advocate - Agency role wherein the principal orientation is to challenge use of natural resources (especially public resources) for profit (especially private benefit) or economic progress philosophies.

Age group - A cohort of organisms of the same age, born within the same year. See also, year class.

Aggradation - A state of channel disequilibrium, whereby the supply of sediment exceeds the transport capacity of the stream, resulting in deposition and storage of sediment in the active channel.

Alluvial channel - A channel that is eroded into sedimentary materials that were previously deposited by the stream under contemporary conditions of flow regime and sediment input.

Annual flow - The total volume of water passing a given point in 1 year. May be expressed as a volume (e.g., acre-feet) or as an equivalent average discharge over the year (e.g., cubic feet per second, cfs).

Arbitrator - Organizations with the statutory authority to establish and enforce management plans or regulations.

Area, cross-sectional - The area of a surface defined by the space between the water surface and the streambed along a transect across the stream, approximated by:

$$A = \sum w_i \, d_i$$

where w_i and d_i are widths and depths of small rectangular sections along the transect.

Area, drainage - The surface area tributary to a lake or stream. Also known as catchment area, watershed area, or river basin area.

Armoring - The process of continually winnowing away smaller substrate materials and leaving a veneer of larger ones.

Backwater - A pool surface created in an upstream direction as a result of the damming effect of a vertical or horizontal channel constriction which impedes the free flow of water.

Base flow - Streamflow contributed solely from shallow groundwater in the absence of significant precipitation or runoff events.

Base level - The lowest elevation to which a stream can erode its bed (e.g., mean sea level is the ultimate base level).

Baseline - A reference condition, against which alternatives are compared (e.g., a hydrologic baseline refers to the current flow regime with all existing water uses in place).

Bias - Systematic under- or overestimation of a parameter resulting from non-representative sampling.

Biological periodicity - The timing of various migratory, reproductive, activity, and growth phases of an animal during the completion of its life cycle.

Brokers - Agencies that have the capability to facilitate natural resources bargaining, especially gifted in distributive politics.

Carrying capacity - The maximum number or biomass of organisms of a given species that can be sustained during a period of least available resources.

Category I criteria - Habitat suitability criteria developed from professional opinion and experience, synthesis from literature, or through negotiated definitions.

Category II criteria - Habitat suitability criteria developed by observing microhabitat conditions occupied by a target organism engaged in a specific activity (e.g., spawning, resting, feeding). Also known as utilization criteria because it does not account for habitat availability.

Category III criteria - Habitat suitability criteria developed by observing used, unused, and/or available microhabitat conditions for a target organism engaged in a specific activity. Also known as electivity or preference criteria because habitat availability is accounted for.

Channelization - Mechanical alteration of a natural stream by dredging, straightening, lining, or other means of accelerating the flow of water.

CFS - Cubic feet per second (measure of streamflow or discharge).

CMS - Cubic meters per second (measure of streamflow or discharge).

Cognitive dissonance - The tendency for people to resist ideas or information that conflicts with their personal world-view and value system.

Cohort table - A matrix showing the numerical abundance or biomass, classified by age groups, for consecutive year classes of animals (or, less frequently, plants) over a period of time.

Colluvium - Material washed onto a floodplain from valley sides or otherwise deposited in a channel by forces other than a stream under its current flow regime (e.g., glacial deposits).

Competition - Active demand by two or more organisms or species for the same environmental resources in excess of the available supply.

Competitive exclusion - Competition resulting in the elimination or extinction of less effectual organisms from a particular ecological niche.

Competitive negotiation - A value-driven negotiation characterized by extreme positional bargaining, wherein concessions are made grudgingly, if at all.

Composite suitability - A weighting factor depicting habitat quality, derived by mathematically aggregating several univariate suitability functions (e.g., by multiplication of univariate suitabilities).

Conditional criteria - Habitat suitability criteria that are designed to simulate differential habitat selection related to the presence or absence of certain conditions (e.g., substitution of large depths as a form of overhead cover).

Conspecific - Of the same species.

Contagion - A habitat metric from the realm of landscape ecology meaning a measure of the clumpiness of patch distributions in a landscape.

Cover - Structural features (e.g., boulders, log jams) or hydraulic characteristics (e.g., turbulence, depth) that provide shelter from currents, energetically efficient feeding stations, and/or visual isolation from competitors or predators.

Cross-section - A plane across a stream channel perpendicular to the direction of water flow.

Datum - A point, line, or surface used as a reference in surveying, mapping, or geology. In IFIM, a datum usually refers to a known or assumed elevation.

Dead storage - That portion of the total capacity of a reservoir reserved for the accumulation of sediment.

Degradation - Erosion and downcutting of an alluvial channel caused when the sediment transport capacity of the stream exceeds the sediment yield from the watershed.

Deterministic - A system with fixed, specified states or regular patterns.

Differential leveling - A surveying technique by which the elevation of a topographic feature is determined by measuring the distance between the unknown point and a horizontal line of known elevation (e.g., the height of the surveying instrument above a datum).

Discharge - The rate of streamflow or the volume of water flowing at a location within a specified time interval. Usually expressed as cubic meters per second (cms) or cubic feet per second (cfs).

Distributive politics - Policy-making arena wherein all legitimate parties are entitled to a "fair share" of the benefits derived from a decision.

Duration - 1. The percentage of time a class of events occurs. 2. An event's time span.

Dynamic equilibrium - A quasi steady-state condition attained in an alluvial channel, whereby sediment supplies are just balanced by sediment transport capacity, resulting in no net change in average streambed elevation over time.

Effective evaporation - The difference between total precipitation and evaporation for a given location.

Effective habitat - The amount of available physical habitat required to accommodate a life stage of a species at its average carrying capacity density, based on average survival and growth rates from a previous life stage.

Effective microhabitat - Physical microhabitat available to organisms having limited mobility, under conditions of rapidly varying streamflow.

Electivity - A mathematical index intended to demonstrate the disproportionate use of a resource with respect to its availability.

Embeddedness - The degree to which the interstitial spaces between substrate materials are filled with fine particles such as silt and sand.

Energy slope - The difference in total energy (potential plus kinetic) of a fluid between two points, divided by the linear distance between the two points.

Evapotranspiration - The combined loss of water from open-surface evaporation and the transpiration of water from leaf and stem tissues of growing vegetation.

Exceedance probability - The probability that an event in a time series will be equaled or exceeded in magnitude by other events in the same series.

Feeding station - 1. A microhabitat type that provides conditions for obtaining large amounts of food with a minimal expenditure of energy. 2. Microhabitat that simultaneously maximizes feeding efficiency and minimizes predation risk.

Fingerling - A small fish, usually in its first year of life (young-of-the-year), but near the end of the first growing season. See also, fry.

Firm yield - A volume of water that can be guaranteed (usually from a reservoir) at a given level of certainty or probability. Less often, firm yield may refer to power production or habitat availability.

Flow regime - The distribution of annual surface runoff from a watershed over time (hours, days, or months). See also, hydrologic regime.

Flushing flow - A stream discharge with sufficient power to remove silt and sand from a gravel/cobble substrate but not enough power to remove gravels.

Froude number - An index of hydraulic turbulence found as the ratio between inertial forces and gravity forces, defined as:

$$F = \frac{V}{\sqrt{gD}}$$

where V is velocity, g is the acceleration of gravity (32 ft/s^2), and D is depth. If F is less than unity, flow is subcritical and described as tranquil or streaming. If F is greater than unity, flow is supercritical and described as torrential or shooting.

Fry - A young, recently hatched fish, early in its first growing season. The life stage that occurs between absorption of the yolk sac (larvae) and complete skeletal development (fingerling).

Geometric mean - 1. In general, a method of calculating the mean of a series of positive numbers as the n^{th} root of the product of n values. 2. In PHABSIM, an alternative algorithm for calculating the composite suitability index from three univariate suitability functions by the equation:

$$csi = (si_d * si_v * si_{ci})^{1/3}$$

Guardians - 1. Groups that view economic progress and development of natural resources for personal or public profit as a positive societal value. 2. Groups that resist change in management practice or design of projects.

Habitat - The physical and biological surroundings in which an organism or biological population usually lives, grows, and reproduces.

Habitat bottleneck - An episode of limited habitat that affects the abundance, growth, and survival of a life stage and is evident in subsequent age groups.

Habitat suitability criteria - Graphical or statistical models that depict the relative utility of increments or classes of macro- or microhabitat variables to a life stage of a target species.

Habitat use guild - Groups of species that share common characteristics of microhabitat use and selection at various stages in their life histories.

Headwater - The source for a stream in the upper tributaries of a drainage basin.

Heteroscedasticity - A phenomenon whereby model prediction errors increase as a function of the magnitude of the independent variable and the errors are equitably distributed between positive and negative.

Historical mitigation - A special application of a habitat time series, whereby the baselines are intended to represent predevelopment conditions.

Hydraulic control - A horizontal or vertical constriction in the channel, such as the crest of a riffle, that creates a backwater effect.

Hydraulic head - The difference in total energy (potential plus kinetic) of a fluid between two points, expressed in units of elevation.

Hydraulic radius - A variable used in hydraulic simulation models, calculated as the ratio between cross-sectional area and wetted perimeter.

Hydrologic cycle - The circulation of water from the ocean to the atmosphere to the land and back to the ocean.

Hydrologic regime - The distribution over time of water in a catchment, among precipitation, evaporation, soil moisture, groundwater storage, surface storage, and runoff.

Hydrograph - A graph showing the variation in discharge over time.

Hydropeaking - The practice of abruptly alternating between a low base and a high peak flow, for electrical power generation during periods of high demand.

Hypolimnion - The lower, colder portion of a lake, separated from the upper warmer portion (epilimnion) by a thermocline.

Incrementalism - The tendency of institutions to follow precedent to make decisions or perform functions. New problems will be resolved with decisions only slightly different from previous decisions.

Index of biotic integrity - A numerical gauge of the biological health of stream fish communities, based on various attributes of species richness, species composition, trophic relations, and fish abundance and condition.

Interflow - Subsurface runoff to an open channel or overland flow that occurs when the rate of soil infiltration exceeds the rate of shallow groundwater recharge immediately following a precipitation event.

Interspersion - A measure of the degree to which patch types in an area are dispersed or fragmented.

Life stage - An arbitrary age classification of an organism into categories related to body morphology and reproductive potential (e.g., spawning, larvae, fry, juvenile, adult).

Longitudinal profile - A plot of an environmental attribute (e.g., elevation, temperature, dissolved oxygen) versus distance along a river under steady-state conditions. Usually expressed with 0 distance at the headwaters or arbitrary upstream starting point and moving downstream.

Longitudinal succession - Gradation in the composition of aquatic communities, lengthwise along a river system from headwaters to mouth.

Macrohabitat - The set of abiotic conditions that control the longitudinal distribution of organisms along one of several environmental gradients: hydrology, geomorphology, temperature, water quality, or energy source.

Manning's n - An empirical calibration parameter used in the Manning equation to represent roughness, or resistance to flow, as a function of the size and irregularity of streambed materials relative to depth of streamflow (e.g., large particles in shallow water are "rougher" than small particles in deep water).

Mesohabitat - A discrete area of stream exhibiting relatively similar characteristics of depth, velocity, slope, substrate, and cover, and variances thereof (e.g., pools with maximum depth < 5 ft, high gradient riffles, side channel backwaters).

Microhabitat - A subset of mesohabitat defining the spatial attributes (e.g., depth, mean column velocity, cover type, and substrate) of physical locations occupied or

used by a life stage of a target species sometime during its life cycle.

Mitigation - Actions taken to ameliorate adverse effects from an existing or proposed action.

Nose velocity - Current speed (usually) measured near the streambed, presumably at the approximate nose level of benthic-oriented fish or macroinvertebrates.

Period of record - The length of time for which data for an environmental variable have been collected on a regular and continuous basis.

Periodicity shift - A change over time in the variance of values in a time series.

Persistence - Nonrandom association of successive members of a time series (e.g., wet periods tend to follow wet periods and dry periods follow dry periods).

Policy - Purposive action taken by public authorities on behalf of or affecting the public.

Policy analysis - The investigation of policy to determine likely outcomes or to understand how a decision was made.

Precipitation regime - Seasonality, form, and effective amount of precipitation that falls at a location on the earth's surface over time.

Pulse attenuation - The effect whereby the difference in flow magnitude is diminished and spread out over time as one moves downstream from the source of the pulse.

Ramping rate - The rate of change in discharge from base flow to generation flow below a peaking hydroelectric facility.

Recurrence interval - The average time interval between events equalling or exceeding a given magnitude in a time series. (See also exceedance probability.)

Redd - A fish nest, usually referring to one constructed by a salmon or trout.

Regulatory politics - Policy-making arena in which an arbitrator makes a clearcut decision based on facts, circumstances, and legal or institutional precedent.

Representative reach - A length of stream used to represent the microhabitat characteristics of a segment, approximately 10-15 channel widths in length, assumed to contain all of the mesohabitat types of the segment in the same proportions as the segment.

Riffle - A shallow, usually rocky portion of a stream with a steeper gradient than the average for the stream.

Riparian - Pertaining to the banks of a natural watercourse, that is, adjacency to the active channel.

Riparian right - The right, as to fishing or the use of water, of one who owns riparian land.

Rule curves - Time-specific storage goals that govern storage and release schedules in reservoirs, based on current storage and projected supplies and demands of water.

Segment - Terminology from IFIM meaning 1. A relatively long (e.g., hundreds of channel widths) section of a river, exhibiting relatively homogeneous conditions of hydrology, channel geomorphology, and pattern. 2. The fundamental accounting unit for total habitat.

Sinuosity - A measure of channel pattern pertaining to the relative amount of meandering exhibited by a stream. Calculated as the ratio between river length and valley length.

Stage - The distance of the water surface in a river above a known datum (e.g., relative to mean sea level).

Stage of zero flow - The water surface elevation at a cross-section when the discharge is zero. For cross-sections not influenced by backwater effects, the SZF is the same as the lowest elevation on the transect.

Standard setting - A policy of using a fixed rule or equation to determine minimum instream flow for a stream, usually based on a hydrological statistic rather than on biological criteria.

Standing crop - The quantity of living organisms present in the environment at a given time, usually expressed as a numerical or biomass density (e.g., kilograms per hectare).

Stationarity - Absence of trends, step functions, or changes in periodicity in a time series.

Substrate, substratum - The surface material of the streambed, for example, sand, gravel, cobble, boulders.

Suitability - A generic term used in IFIM to indicate the relative quality of a range of environmental conditions for a target species.

Superimposed stream - A stream with a deeply incised channel, often cut into bedrock, that is formed when the rock strata underlying a previously existing stream channel are slowly uplifted.

Surface runoff - Overland flow of water that occurs when the rate of precipitation exceeds the rate of infiltration of water into the soil moisture zone.

Synergism - Two or more substances or factors acting together to achieve an effect of which each part is individually incapable.

Terrace - An alluvial feature of streams formed by downcutting and subsequent abandonment of a former floodplain, with the development of a new floodplain within the walls of the escarpment.

Thalweg - 1. A longitudinal profile of the lowest elevations of a sequential series of cross sections. 2. The lowest elevation at an individual cross section.

Time step - The interval over which elements in a time series are averaged.

Topographic elevation - The angle between horizontal (0°) and the average height of the topographic horizon, measured from the middle of the channel.

Total habitat - The total area of habitat available to a life stage or species in a river segment, found by integrating the length of stream having suitable macrohabitat conditions with the unit microhabitat at a given discharge.

Transferability - 1. Applicability of a model (e.g., habitat suitability criteria) to settings or conditions that differ from the setting or conditions under which the model was developed. 2. Applicability of data obtained from a remote source (e.g., a meteorological station) for use at a location having different environmental attributes.

Trend - A unidirectional change in the average values over time of members of a time series.

Trigger levels - Operational guidelines that tell a reservoir operator when to switch from one rule curve to another.

Type I error - Error of rejecting a true null hypothesis. In criteria transferability tests, the error of accepting non-transferable criteria.

Type II error - Error of accepting a false null hypothesis. In criteria transferability tests, the error of rejecting perfectly good criteria.

Utilization curve - A univariate habitat suitability index curve that was derived from observations and measurements of locations occupied by the target species. No correction or adjustment for habitat availability is made for a utilization curve.

Variable backwater - A pooling effect in a tributary mouth or a hydraulically connected side channel caused when the water surface elevation in the main channel is higher than the normal depth of the tributary or side channel. As the main channel water surface subsides, the pooling effect disappears or becomes much less influential.

Variable voltage pulsator - An electronic device that converts alternating current to pulsed direct current, for use in electrofishing.

Vegetative offset - The average distance from the edge of a stream to the trunks of riparian trees.

Velocity adjustment factor - A coefficient derived in the IFG4 hydraulic simulation program as the ratio between the input discharge and the discharge calculated from initial estimates of hydraulic variables, used to achieve mass balance between input and predicted discharges.

Vertical - A location along a transect across a river, where microhabitat-related data are collected.

Water year - The period from 1 October through 30 September, usually considered to represent the annual hydrologic cycle beginning when the natural discharge typically approaches base flow in North America and Europe.

Water yield - The total annual surface runoff from a watershed, measured as open channel flow.

Wetted perimeter - The length of the line of intersection of the channel wetted surface with a cross-sectional plane normal to the direction of flow. Approximately equal to the wetted width plus twice the average depth of flow.

Year class - A cohort of organisms born in the same calendar year.

Appendix. Quantiles of the Hotelling-Pabst Test Statistic for α = 0.05. Adapted from Conover (1980).

n	$w_{0.025}$	$w_{0.975}$	n[1]	$w_{0.025}$	$w_{0.975}$
10	60	273	30	2,868	6,132
11	86	357	31	3,185	6,745
12	120	456	32	3,535	7,387
13	162	570	33	3,911	8,068
14	212	702	34	4,312	8,789
15	270	855	35	4,740	9,551
16	340	1,025	36	5,196	10,356
17	420	1,217	37	5,680	11,204
18	512	1,432	38	6,194	12,096
19	618	1,668	39	6,739	13,034
20	738	1,928	40	7,314	14,019
21	870	2,217	41	7,922	15,051
22	1,020	2,529	42	8,563	16,132
23	1,184	2,871	43	9,239	17,263
24	1,366	3,242	44	9,949	18,446
25	1,566	3,642	45	10,695	19,680
26	1,786	4,072	46	11,477	20,968
27	2,024	4,537	47	12,298	22,310
28	2,284	5,033	48	13,157	23,707
29	2,564	5,565	49	14,055	25,161

[1]For n greater than 30, lower quantiles ($w_{0.025}$) were estimated by:

$$w_{0.025} = \frac{n(n^2-1)}{6} + x_{0.025}\frac{n(n^2-1)}{6\sqrt{n-1}}$$

where $x_{0.025}$ is the 0.025 quantile of a standard normal random variable. Upper quantiles ($w_{0.975}$) were found by :

$$w_{0.975} = \frac{n(n^2-1) - w_{0.025}}{3}$$

Both equations and instructions for use were provided by Conover (1980).

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